

10.2 CHEMICAL-SPECIFIC CONCEPTUAL SITE MODELS

CSMs for the 13 indicator contaminants ~~ICs~~ are presented in the following subsections. Each discussion includes with a brief presentation of contaminant distribution, potential sources and pathways, and loading, fate, and transport, and concludes with a CSM summary. For each CSM, a three-section panel series is provided that presents cross-media contaminant distributions and available source information and presents a subset of the contaminant distribution data. An electronic version of the three-section CSM panels is included in Appendix I.

Panel A presents summary information on the observed concentration in surface sediment using Thiessen polygons to spatially represent concentrations between data points, sediment traps, riparian soil/sediment, surface water), and TZW from RM 1.9 to 11.8, excluding dredge and cap sample locations. A histogram of the surface sediment data is included in the top left-hand corner of each panel to show the distribution of the data. Unfiltered push probe, filtered push probe, and peeper results are displayed for TZW. Surface water XAD data are presented for total PCBs, dioxins/furans, total DDx, total PAHs, total chlordanes, aldrin, and dieldrin. Surface water peristaltic pump data are presented for arsenic, chromium, copper, zinc, and TBT. The BEHP data presented are a combination of the XAD and peristaltic pump data.

Panel B presents a summary of subsurface sediment concentrations and large-scale (>30 cm) erosional/depositional areas predicted for a major flood based on the FS HST model for the Study Area (Chen 2011, pers. comm.). Thiessen polygons on these panels represent concentrations in the sediment interval just below the surface sediment (typically the B interval). A histogram of the subsurface sediment data is included in the top left-hand corner of each panel. Also included are icons depicting the locations of 10 major types of historical industries that are or were active in the Study Area.

In addition to the erosional/depositional information shown on Panel B, Map 10.2-1 shows areas in the Study Area at risk for surface sediment disturbance from incidental anthropogenic activities based on water depth and in-water operations. This includes all areas above the -5 ft NAVD88 contour that are potentially subject to boat wakes, areas in the immediate vicinity of docks and berths, and any additional areas where sediment scour that did not appear to be due to natural forces was evident in the 2002–2009 bathymetric time-series data set.¹ This map does not include an analysis of those activities that are specifically intended to move sediments (shoreline/structure construction, maintenance dredging, or remedial capping/dredging).

Panel C presents whole-body concentrations in field-collected smallmouth bass, clams, crayfish, and sculpin. A detailed view of the composite groupings can be found on Map 2.2-10a–d.

¹ Map 10.2-1 is a qualitative presentation of areas where there is a reason to believe that anthropogenic disturbance risk may be relatively higher than other areas.

For each upland site that has undergone sufficient investigation to identify known or likely complete pathways, a box is shown on the panels listing the applicable pathways and noting whether they are complete or likely complete. Where there are insufficient data to make a determination or when a complete pathway was determined to be not present (see Tables 10.2-1 and 10.2-3 through 10.2-14), no information is presented.

As noted in Section 4.2, the source information presented here is highly dependent upon whether a site is involved in DEQ's cleanup program and the degree of investigation and data generation². As shown on Table 4.2-1, several sites adjacent to or near the Study Area are not in the cleanup program, and it is likely that many sites, particularly those that are the location of historical facilities that operated outside the boundaries of current sites, are not fully addressed in DEQ files. As a result, this section does not represent a complete inventory of sites and operations that contribute or have contributed to contamination in Portland Harbor. These limitations on source information primarily affect historical sources. The understanding of current sources may also be affected by these limitations; however, available information appears to be adequate for the purposes of the FS.

Along with the panels, three figures are provided for each CSM chemical to portray loading, fate, and transport processes under current conditions in the Study Area³. The first figure consists of a pair of box and whisker plots; the first compares the range of the estimated external and internal annual loads to the Study Area for each of the loading terms quantified for a given CSM contaminant, while the second compares the concentrations of the chemical in surface sediment, sediment traps, and suspended solids in surface water for the entire Study Area. The second figure is a box-and-arrow diagram⁴ depicting relevant loading, fate, and transport processes for each CSM chemical at the Study Area scale. The third figure provides a graphical comparison, by river mile, of the quantified external and internal loading terms, including central, upper, and lower estimates, and affords additional resolution of spatial patterns in loading to the Study Area. As documented in Section 6.1 and Appendix E, external loading of each CSM contaminant to the Study Area was estimated quantitatively ~~or~~ semi-quantitatively for upstream surface water,⁵ stormwater, atmospheric deposition to

Commented [Int1]: For consistency with revised Section 6 terminology. Section 10 Box/Arrow Figure labels need to be edited.

² The source information presented in this Portland Harbor RI report is a compilation of public information available from site owners and operators and from DEQ, and is based upon information provided through September 2010, and DEQ's September 2010 Source Control Milestone Report. Source information will be updated in the Portland Harbor FS report. For the most up-to-date DEQ source information, DEQ's November 2014 Source Control Milestone Report is available online at <http://www.deq.state.or.us/lq/cu/nwr/portlandharbor/report.html>

³ Includes surface water and bedded sediments in the surface mixed layer (0–40 cm bml).

⁴ This diagram does not attempt a mass balance because sufficient data are not available and because of the varying levels of quantification (qualitative to quantitative) of each term.

⁵ As discussed in Section 6.1, estimated upstream surface water loads were developed using data from both RM 11 and RM 16. Because of the complex hydrodynamics on the LWR between its confluence with the Columbia River (RM 0) and the entrance to Multnomah Channel at RM 3 (frequent flow reversals, see Section 3.1.3.3), surface water chemical loads leaving the Study Area at RM 1.9 could not be estimated using the simplified approach described in Section 6.1. The furthest downstream surface water loads for the LWR were estimated at RM 4. Surface water loads exiting the Study Area via Multnomah Channel were also estimated.

the water surface, and groundwater advection through subsurface sediments. Quantitative ~~to semi-quantitative~~ estimates were also generated for upland groundwater plumes and permitted point-source discharges for a subset of the CSM contaminants for which these terms may be significant. Unquantified loading terms, including bedload, volatilization, and riverbank erosion, are represented qualitatively on the box-and-arrow diagrams. The only contaminant fate and transport mechanism internal to the Study Area for which quantitative estimates were developed in the RI is pore water advection from surface sediment to the overlying surface water column. Other internal fate and transport mechanisms, including sediment erosion, sediment deposition, sediment burial, and biological and geochemical transformation (degradation) are represented qualitatively on the box-and-arrow diagrams.

10.2.1 Total PCBs

The Study Area graphical CSM for PCBs is presented on Panels 10.2-1A–C. ~~The data from recent sediment studies at RM 11E and in the downtown reach are included in the updated RI data set and presented in Appendix H. These data are not included on Panels 10.2-1A–C, but are considered in the narrative discussion below as part of the revised CSM for PCBs.~~

Commented [Int2]: The panels have been updated to include data formerly included in Appendix H in the 2011 Draft Final RI.

PCBs are a class of nonpolar, synthetic, halogenated hydrocarbons that were manufactured in the United States between 1929 and 1977 and widely used for a variety of purposes.⁶ Current allowed uses include transformers, heat transfer systems, natural gas pipelines, existing carbonless copy paper, and electrical switches. Historical PCB uses included dielectric fluids in transformers and capacitors, electrical cables, cutting oils, hydraulic oils, lubricants, heat transfer fluids, plasticizers, flame retardants, additives to pesticides, paints, carbonless copy paper, caulk, adhesives, sealants, in heat transfer systems, electromagnets, and for dust suppression. PCB sources in waste materials include scrap metal recycling, auto salvage, used oil, recycled paper, asphalt roofing materials, building demolition, and in the repair and salvaging of ships, locomotives, heavy equipment, and manufacturing equipment. Although PCBs are ubiquitous in the environment, commercial PCB production in the United States ended in 1977.

~~Although PCBs do not readily degrade in the environment (e.g., by reductive dehalogenation), they are persistent and are lipophilic, persistent, and tend to bioaccumulate (Erickson 1997). They are hydrophobic/lipophilic organic substances that accumulate in organisms both by uptake from the environment over time (bioaccumulation) and along the food chain (Erickson 1997). PCBs biomagnify with each trophic level in the food web (biomagnification). In aquatic organisms, the rate and physiological mechanism of PCB metabolism depend on the species and the specific type of PCB.~~

~~See DEQ (2003a) for additional information on PCB sources.~~

As PCBs are no longer produced and sold, significant primary releases to the environment are uncommon; however, relatively small PCB releases still occur. Consequently, most of the mass of PCBs found in the Study Area sediments is derived from historical sources. In addition, secondary sources may introduce PCBs to the LWR through a variety of environmental pathways as described in Section 10.2.1.2.

Commented [Int3]: This was deleted here but this concept was added to the PCB CSM Summary section.

10.2.1.1 PCB Contaminant Distribution

For the purposes of this discussion, PCB concentrations greater than 300 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 400 pg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-1A-C).

Commented [Int4]: These definitions were deleted for some IC and not for other, we re-inserted them for all.

On a harbor-wide basis, the highest PCB sediment concentrations occur in depositional nearshore areas and in locations proximal to local upland sources (Map 5.1-2a-m and Panels 10.2-1A-B). Relatively high concentrations of PCBs are also often found in riparian sediments, sediment trap samples, surface waters, and biota samples in the areas with elevated sediment concentrations.

Similar spatial and concentration trends are observed for subsurface sediments (Panel 10.2-1B). Areas where surface and subsurface concentrations are not well correlated may be an indication of spatially and temporally variable inputs and sources, or to different influences from sediment transport mechanisms. Areas where the highest concentrations of PCBs in sediment are observed include RM 11.3E, RM 98.8-10W, Swan Island Lagoon, International Slip (RM 4E3.7-3.8E), RM 2.1-2.5E, and RM 4.0 to 4.1E. Total PCB concentrations are generally higher in subsurface sediments (Panels 10.2-1A-B, Maps 5.1-2a-m, and Figure 5.1-33), pointing to predominantly historical total PCB sources and higher past loads. but eExceptions to this general trend are noted at RM 11E, Swan Island Lagoon, and Willamette Cove.

Relatively high concentrations of PCBs in surface water are generally found in areas with elevated sediment concentrations. The highest total PCB concentrations were associated with single point samples in Willamette Cove and at RM 5.5. Total PCB concentrations at RM 11 were consistently higher than at RM 16, suggesting a source or sources between these locations. Aside from these exceptions, the range of total PCB concentrations within the Study Area surface water was fairly consistent.

In the sediment trap data, PCB concentrations were generally higher in samples collected between RM 6.7 and 11.3 compared to other locations. Except for large differences in PCB levels in sediment traps between ST007 (RM 11.3E) and its paired location ST008 (RM 11.5W), there was little relative difference in PCB concentrations between other cross-river sample pairs, suggesting suspended sediments are laterally homogenized in most of the Study Area. Total PCB congener concentrations in the Study Area samples were all higher than the average PCB concentrations from upstream locations.

~~The H~~igh concentrations of PCBs were typically found in biota samples in areas with elevated sediment concentrations.

Spatial variations in PCB composition (based on congener data) are evident throughout the Study Area, and areas of elevated PCB sediment concentrations often exhibit congener homolog patterns that are distinct from surrounding areas of lower PCB concentrations (Maps 5.1-32 and 5.1-33; Figures 5.1-35a–c and 5.1-36a–c). In general, these areas of lower concentrations are predominated by similar proportions of the tetraCB, pentaCB, hexaCB, and heptaCB homolog groups, with some localized exceptions. Areas of higher concentration (greater than 1,000 µg/kg) tend to be predominated by two or three of these homolog groups and may have a higher proportion of other homolog groups.

Commented [Int5]: Discussion of chemical signatures and chemical group composition should be retained. This information is important to the CSM (in particular, aids in interpretation of single vs. multiple sources).

For the areas of elevated concentration noted above, the PCB homolog patterns in surface and subsurface sediment, as well as in the sediment traps and in the particulate portion of the surface water samples, are often similar within each area. Subsurface sediment patterns are less consistent with surface sediment homolog patterns for the areas at RM 6.9 to 7.5W and RM 2.1 to 2.5E compared with the other elevated areas. Across elevated areas, homolog distributions vary spatially between areas, further suggesting multiple sources of PCBs for these areas of elevated concentration.

Relatively low levels of PCBs are widespread in portions of the harbor away from the localized areas of elevated concentrations (Panel 10.2-1A and Map 5.1-1). The histogram inset on Panel 10.2-1A shows that 57 percent of the detected surface sediment results have total PCB concentrations less than or equal to 34 µg/kg. The homolog patterns in these widespread, low level PCB areas are generally similar in both depositional and erosional areas, have less distinct variations than areas of higher concentration, and may reflect inputs from upstream and transport within the Study Area.

10.2.1.2 Potential PCB Sources and Pathways

Numerous upland sites have been identified as being known or likely historical and/or current sources of PCBs. These sites discharge directly to the river or discharge through shared conveyance systems (Table 4.4-1). Historical and current known or likely complete pathways for PCBs in stormwater have been identified at several properties associated with former shipyards and sites where transformers were serviced and/or dismantled. Historical wastewater discharges associated with ship building and decommissioning, electrical component manufacturing, and leaks and spills from equipment that used PCB-containing fluids s (e.g., hydraulic oils) are likely, but have not been specifically identified or quantified. Wastewater discharges are currently regulated primarily through NPDES permits.

Potential upland and overwater sources and identified known and likely complete migration pathways are identified on Table 10.2-1 and Panels 10.2-1A–C. These sources and pathways, identified on the basis of the process described in Section 4.2,

focus on ECSI sites and are based on a review of information in the associated DEQ ECSI files and other readily available site information, including, in the case of LWG-member sites, information provided by the site owner.

The most significant migration pathways for PCBs in the Study Area are historical and included industrial wastewater, stormwater, overland transport, overwater releases, and riverbank erosion (Table 10.2-1). Atmospheric deposition and upstream inputs may have also contributed PCBs to the Study Area, the releases from these pathways are not quantifiable and are difficult to distinguish.

Stormwater PCB loads have decreased substantially from historical levels since implementation of stormwater controls and the statutory ban on PCB manufacture in the 1970s. Overland transport was likely more important prior to the development of extensive stormwater conveyance systems. Bank erosion is also likely more important when PCBs were in wider use, or when contaminated material was used in construction fill activities. Historical PCB overwater releases have not been identified through the file review process, but are likely to have occurred in association with overwater operations, such as ship building and dismantling, ship repair and maintenance, and with the use of hydraulic fluids in dock operations. The locations of elevated PCB concentrations in sediments coincide in some cases with ship construction, dismantling, and repair operations, and it is likely that overwater releases occurred concurrently as a result of historical activities in these locations. PCBs are also detected in sediments near outfalls draining facilities historically engaged in electrical equipment manufacturing, such as at RM 11.3E and at OF-17. Stormwater discharges and riverbank erosion associated with fill soil from offsite and/or steel manufacturing activities at RM 2E have also resulted in PCB contamination in sediments.

Current PCB inputs to the Study Area are much lower than historical inputs. However, measured elevated levels of PCB concentrations in surface sediments and other media, including biota in the International Slip and Swan Island Lagoon indicate ongoing localized inputs and/or internal mass transfer of historical PCB inventory from subsurface to surface sediments and then to other media by processes such as sediment resuspension (due to both natural and anthropogenic disturbance factors) and biological uptake. While surface sediments generally exhibit lower PCB concentrations than subsurface sediments, the temporal persistence of elevated PCB levels in surface sediments in many nearshore and off-channel areas suggests that net sedimentation rates may be low in many nearshore areas. This is supported by the bathymetric change data and the limited radioisotope data from the Study Area (Anchor 2005), sediment column mixing rates are high, and inputs of PCBs still occur. Potentially important current pathways include stormwater and riverbank erosion. The effects of current releases and these physical site features are expressed in surface sediment concentrations (Panel 10.2-1A).

Known or likely complete historical pathways for PCBs have been identified at 40 sites (Table 10.2-1 and Panels 10.2-1A–C), and include stormwater (38 sites), overland transport and riverbank erosion (6 sites), and overwater releases (1 site).

Current known or likely complete pathways for PCBs have been identified at 18 sites, and include stormwater (16 sites), overland transport (2 sites), riverbank erosion (3 sites,) and groundwater transport (1 site).

10.2.1.3 Loading, Fate, and Transport of PCBs

PCB loading, fate, and transport in a typical year for the Study Area is summarized on Figures 10.2-1a-b, 10.2-2, and 10.2-3 ~~summarize the~~. Estimates are for current conditions and likely differed historically. Much of the PCB mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

External Loading

Upstream surface water represents the largest estimated current loading term for PCBs to the Study Area (Figures 10.2-1a and 10.2-2), and is associated with both the dissolved and suspended fractions. On an annually averaged basis, the majority of this load occurs during low-flow conditions (Figure 6.1-2) which exist for approximately two-thirds of the year. Surface water samples collected during high-flow consistently exhibited lower concentrations of PCBs than in low-flow samples, indicating that inflow concentrations at high flow rates have greater influence than local effects. Total PCB concentrations in sediments accumulating in upstream borrow pits, which are likely a mixture of upstream bedload and suspended load, are comparable to the upriver bedded sediment background concentrations.

PCB loads in surface water increase between the upstream and downstream boundaries of the Study Area in both the particulate and dissolved fractions, and is in part attributable to quantified external loads (stormwater, atmospheric deposition, and advection through subsurface sediments). Other possible reasons for the increased loading is due to internal fate and transport processes such as sediment resuspension, which have not been quantified. The distribution of total PCBs in surface sediments, sediment trap samples, and the particulate fraction of surface water samples, on both a dry-weight and OC-normalized basis, is presented on Figure 10.2-1b.

Estimates of current PCB loading via stormwater are approximately half the estimated upstream load, atmospheric deposition directly to the Study Area river surface (is nearly an order of magnitude lower than the upstream surface water load (Figure 10.2-2). PCBs were detected in stormwater in Round 3A and 3B sampling in each land use area sampled (see Section 6.1.2.3 and 10.1.4.1.2). Groundwater advection through subsurface sediments is estimated the least significant of the quantified terms, but is subject to a relatively high degree of uncertainty due to the variability in published organic-carbon partitioning values for PCBs. As discussed in Section 6.1.1.2, bedload into- and out of the Study Area is expected to be low relative to dissolved and

particulate surface water loading. PCB volatilization from the water column is relevant for only a small fraction of the less chlorinated PCB congeners and is also expected to be low.

Fate and transport processes internal to the Study Area for total PCBs are shown on Figure 10.2-2. PCB transport to the water column due to pore water advection through surface sediments is only process for which quantitative estimates were developed, and is estimated to be similar in magnitude to the subsurface advective loading. Other internal fate and transport processes are depicted on Figure 10.2-2 on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rate. PCB partitioning between suspended sediment and surface water depends on the relative concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. The fate of PCBs within the Study Area may also be influenced ~~to a minor degree by~~ relatively ~~very~~ slow microbially-facilitated degradation and photolysis.

PCB loads from upland groundwater plumes are not expected to be significant and estimates were not generated. Loading from permitted point source discharges were not estimated because PCBs are not regulated and monitored under any active discharge permits within the Study Area. Estimates of PCB loading from upland soil and riverbank erosion also were not assessed due to a paucity of data for riparian soil PCB concentrations and erosion rates.

The total PCB load in surface water increases downstream through the Study Area to RM 4⁷ (Figure 10.2-3), the largest PCB stormwater inputs enter the Study Area between RM 3 and 4. As described in Section 6.1.2, the estimated load (1 kg/yr) to this reach is largely from a non-representative (unique) site (Outfall WR-384) and exceeds the next highest stormwater load, between RM 8 and 9, by a factor of 10.

Atmospheric deposition is estimated to contribute a total PCB load approximately one-third that of stormwater at the Study Area scale. Deposition loading to the water surface varies only as a function of water surface area by river mile.

No current known or likely complete PCB overwater pathways have been identified. Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall.

10.2.1.4 PCB CSM Summary

PCBs are no longer produced; however, they are common in the environment due to widespread historical sources, secondary releases, and their environmental persistence. PCBs are present in Portland Harbor surface and subsurface sediments, other abiotic media (such as surface water), and fish and shellfish tissues. PCB concentrations in

⁷ At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3.3).

sediments across much of the Study Area range from nondetected to about 34 µg/kg. This is about # times the upriver background level of XX µg/kg corrected for the organic carbon content. In contrast, a number of distinct subareas of the Study Area exhibit surface and/or subsurface sediment total PCB concentrations that greatly exceed these levels. With minor exceptions, these subareas are located in nearshore (outside of the navigation channel) and off-channel (off the mainstem of the river, such as Swan Island Lagoon) areas that are both more quiescent and proximal to likely or known historical sources. PCB composition (homolog patterns) shows variations between these subareas, suggesting multiple localized sources with minimal subsequent in-river transport. The largest areas of elevated PCBs in sediment include the areas offshore of the former Albina Shipyard at RM 11.3, Gunderson, Swan Island Lagoon, International Slip, RM 4E (International Terminal), and EOSM.

Commented [Int6]: Value to be added following dispute resolution

In most of the subareas with elevated PCB sediment concentrations, and throughout the Study Area generally, subsurface PCB concentrations exceed surface levels, indicating greater inputs or releases historically that have been reduced or eliminated over time. Small-home-range fish and shellfish tissue samples collocated with these subareas generally exhibit elevated levels, indicating ongoing exposure and biological uptake. The localized long-term accumulation of sediments (and PCBs) in the sediment column, as well as the physical site characteristics (e.g., time-series bathymetric changes) and HST modeling, indicate that most of these subareas are depositional/stable over time.

Noteworthy exceptions to the pattern of higher concentrations at depth are found at RM 11.3E, in Willamette Cove, and in Swan Island Lagoon. Several sampling points in the area of RM 11.3 exhibit higher surface sediment than subsurface levels along the eastern nearshore area and adjacent channel edge. The nearshore sediment PCB distribution, as well as the collocated surface water and sediment trap data, suggest a local, recent input and/or redistribution of PCBs historically released into this area and present in the sediments. Anthropogenic sediment disturbance (i.e., prop wash/dredging) in this area also may have altered the distribution of PCBs with depth in the sediment column and re-exposed and re-mobilized subsurface contamination.

Collocated surface and subsurface samples from the inner portion of Willamette Cove also exhibit higher surface PCB concentrations than at depth which may be indicative of higher recent inputs. Finally, in Swan Island Lagoon mean surface and subsurface total PCBs concentrations are approximately the same. The lack of a vertical gradient may reflect a combination of time-varying inputs, low net sedimentation rates, and localized high surface sediment mixing rates that result in variable spatial trends in sediment quality with depth.

While the elevated PCB subareas in the Study Area appear to result from releases from specific localized sources (both historical and/or current), PCBs also enter the Study Area from non-point and diffuse sources, such as private and public stormwater and sewer outfalls, and sources upstream of the Study Area in the Willamette River. Harbor-wide loading estimates indicate that the highest current external inputs to the

Study Area are from upstream surface water and, to a lesser degree, stormwater runoff and atmospheric deposition within the Study Area. These estimates indicate that the mass flux of PCBs in surface water (dissolved and particulate) exiting the Study Area at RM 2 and at the Multnomah Channel entrance exceed the estimated fluxes entering the Study Area from all quantified external loading terms. This suggests an internal mass transfer of PCBs from bedded sediment to the water column, likely through sediment resuspension.

PCBs are the primary contributor to risk in Portland Harbor. The highest human health risks from exposure to PCB concentrations were associated with the fish consumption pathway, for which PCBs contribute approximately 93 percent of the cumulative cancer risk for whole-body consumption and approximately 97 percent of the cumulative risk for fillet consumption, on a Study Area-wide basis.

Total PCBs pose significant risk to mink, river otter, and spotted sandpiper, and low risk to osprey, bald eagle, sculpin, and smallmouth bass. PCBs were identified as one of the primary contaminants in sediment likely to pose potentially unacceptable risk to the benthic community or populations. Risks to the benthic community are found in localized areas and generally co-occur with other contaminants potentially posing risk. The highest risk estimates are for mammalian predators, represented by mink and river otter.

The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling conducted as part of the FS. Elevated levels of PCBs in tissue are co-vary with elevated levels of sediment contamination, surface water contamination, and sediment trap contamination. Furthermore, elevated surface water contamination seems to be associated with elevated sediment contamination. The reasons for these relationships likely vary. In some cases, surface water contamination may be associated with stormwater discharge, and in other cases it may be associated with bedded sediment. Overall, indications are that PCB levels in surface sediments in most areas are decreasing over time. The rate of this improvement is a function of localized net sedimentation rates, erosion potential, surface sediment mixing rates, in-situ degradation processes, and source control actions.

10.2.2 Total PCDD/Fs

A graphical CSM for Total PCDD/Fs in the Study Area is presented Panels 10.2-2A–C. As a group, PCDDs represent 75 different positional isomers, while PCDFs comprise over 135 compounds (Eisler 1986). These two chemical classes are generally referred to as dioxins and furans, respectively. PCDD/Fs are byproducts of combustion, incineration, certain industrial chemistry processes, and natural sources, including combustion, metal smelting, and production bleached paper, polyvinyl chloride (PVC), inks/dyes, certain chlorine production technologies, chlorophenols, chlorinated herbicides, and commercial Aroclor (PCB) mixtures (ATSDR 1998). Examples of combustion and incineration that may lead to the formation of PCDD/Fs include hazardous or medical waste incinerators, cement kilns, boilers and industrial furnaces,

vehicle emissions, fossil fuel power plants, burning PCB containing electrical equipment (such as transformers), and backyard burning (e.g., refuse piles, burn barrels). PCDD/Fs are naturally produced from forest fires, volcanic eruptions, and sedimentary deposits. Currently the largest source is from backyard burning (EPA 2006b). The primary source of PCDD/Fs in environmental media is generally atmospheric deposition. When released into the air, some PCDD/Fs may be transported long distances, even around the globe. In the atmosphere, it has been estimated that 20 to 60 percent of 2,3,7,8-TCDD is in the vapor phase. Sunlight and atmospheric chemicals can break down a very small portion of the PCDD/Fs, but most PCDD/Fs will be deposited on land or water (ATSDR 1998).

Behavior of PCDD/Fs in the environment is characterized by low vapor pressures, high octanol-water and organic carbon partitioning coefficients (K_{ow} and K_{oc} , respectively), and extremely low water solubilities. These compounds are hydrophobic and factors have indicate a strong affinity for sediments, particularly sediments with high organic matter content; as a result, transport of PCDD/Fs in aquatic systems is closely tied to fine-grained sediment transport processes. Some PCDD/Fs, present near the water surface and not bound to solids, may be subject to photodegradation (EPA 1994). but this is only relevant for a small fraction of the PCDD/Fs in the system. In general, PCDD/Fs are stable in all environmental media, with persistence measured in decades. Chemical degradation of PCDD/Fs through reductive dechlorination can also occur, but it is can be a very slow process. PCDD/Fs have been shown to biomagnify in aquatic food webs and associated avian and mammalian species (ATSDR 1998).

10.2.2.1 ~~TCDD TEQ~~ Total PCDD/F Contaminant Distribution

Panels 10.2-2A–C present the Study Area graphical CSM for ~~TCDD TEQ~~ Total PCDD/F. For ~~TCDD TEQ~~ Total PCDD/Fs, concentrations greater than ~~5-1,000~~ pg/g in sediment, 10 pg/g in biota samples, and greater than 0.25 pg/L in surface water samples are considered elevated for this discussion.

Care should be taken in interpreting ~~the TCDD TEQ~~ Panels 10.2-2A–B because relatively few sediment samples from the federal navigation channel were analyzed for PCDD/Fs. The size of a Thiessen polygon is determined by one-half the distance to the nearest sampling point in any direction, and when no data exist for the federal navigation channel, the polygon is drawn such that the concentration extends across the channel (e.g., see red and yellow surface sediment polygons above RM 11). The full surface and subsurface data sets shown in Maps 5.1-5 and 5.1-6a–m are useful in interpreting these panels.

~~10.2.2.1.1~~ Sediment

~~Total PCDD/Fs~~ were detected in sediments in several locations along the eastern and western nearshore zones and in Swan Island Lagoon. The highest detected concentrations were found in the eastern nearshore zone at RM 2E-8E, Swan Island Lagoon, RM 11E, and in the western nearshore zone from RM 6W-10.3W, and from

Commented [A7]: It would be more relevant to discuss dioxin TEQ. Total PCDD/Fs may be good for relative distribution, but the numbers are a bit meaningless in terms of interpretation since most readers do not use this measurement for dioxins / furans. I suggest adding TEQ concentrations to the text.

Per Discussion with K. Koch on 12/4, this section is now based on Total PCDD/F data. G. Revelas

RM 4W-6W, and at RM 3.4W. While limited surface PCDD/F data are available for the navigation channel, the highest total PCDD/F concentrations are located at RM 6-7 and from RM 11-11.8, and appear to be associated with higher concentrations found in the eastern nearshore zone. Total PCDD/F concentrations in the subsurface are generally greater than that observed in surface sediments. The higher concentrations generally observed in subsurface sediment relative to concentrations in surface sediment are indicative of a primarily historical input of these contaminants to the Study Area.

Surface water concentration are elevated in Willamette Cove, at RM 6.8W, and in the RM 4 transect sample. Total PCDD/F surface water concentrations within the Study Area did not display any consistent trends from upstream to downstream.

There were no strong spatial or temporal gradients evident in total PCDD/F concentrations measured in suspended sediments collected in sediment traps within the Study Area (Figure 5.2-14). In general, Study Area sediment trap samples had higher total PCDD/F concentrations than the upstream locations near RM 15.6.

PCDD/Fs were detected in all fish and invertebrate tissue samples collected from the Study Area, the highest concentrations were observed in samples collected between RM 6.5 and 7.5 (Panel 10.2-2C).

10.2.2.2 Potential PCDD/Fs Sources and Pathways

Historical known complete or likely complete pathways for PCDD/Fs have been identified at four sites (Table 10.2-3 and Panels 10.2-2A–C) and include stormwater (four sites), groundwater (one site), overwater releases (one site), overland transport (two sites), and riverbank erosion (two sites).

Current known complete or likely complete pathways for PCDD/F have been identified at two sites and include stormwater (both sites), overland transport (one site), and riverbank erosion (one site).

Stormwater transport is expected to be the most significant current pathway for PCDD/Fs to enter the Study Area from adjacent upland sites. A current known complete pathway has been identified for Triangle Park and City of Portland Outfall OF-22B Gould Electronics, Rhone Poulenc, and McCormick & Baxter are identified as having known complete historical pathways.

No current known or likely PCDD/F overwater pathways have been identified. However, McCormick & Baxter is identified as having a known historical pathway.

No sites have been identified as having current known or likely complete groundwater pathways for PCDD/Fs, though McCormick & Baxter had a historical complete pathway.

A current known pathway for PCDD/Fs exists at the Triangle Park site. McCormick & Baxter had a historical known complete pathway and Triangle Park had a historical likely complete pathway for PCDD/Fs.

PCDD/Fs were identified as potential contaminants at two upstream paper mills (Blue Heron and West Linn).

10.2.2.3 Loading, Fate, and Transport of PCDD/Fs

The loading, fate, and transport assessment total PCDD/F is summarized on Figures 10.2-4a, 10.2-5a, and 10.2-6a. The greater PCDD/Fs mass in the Study Area deeper sediments (Panel 10.2-2B) is likely attributable to historical loading that occurred under different past loading conditions and rates. Due to the limited data available, PCDD/F loading terms were quantified only for upstream surface water and advection through subsurface sediments. Upstream loading contributes much greater estimated loads of total PCDD/Fs to the Study Area than advection through subsurface sediments (Figures 10.2-4a and 10.2-5a). Upstream surface water loads are associated primarily with the suspended particulate fraction, with similar total PCDD/F mass loads entering the Study Area during high-flow and low-flow conditions (see Figure 6.1-4). The total PCDD/F upstream loads in surface water comparable with downstream loads. A cross-media comparison and statistical assessment of surface sediment, sediment trap samples, and the particulate fraction of surface water samples on a Study Area-wide basis is shown in Tables 10.2-2a-b and Figure 10.2-4b.

10.2.2.4 Relationship of Sources to Distribution of PCDD/Fs CSM Summary

Detected PCDD/F concentrations in most Study Area surface sediments range from 0 to 500 pg/g. This compares to upriver background levels of XX pg/g (OC-equivalent basis). There are a number of distinct subareas scattered throughout the Study Area that exhibit elevated PCDD/F concentrations. One of these subareas is between RM 6.5 and 7.5 along both the east and west banks of the river and in the adjacent channel. These areas coincide with several currently identified known or likely historical industrial dioxin and/or furan sources. In addition, some surface water and biota (bass, sculpin, and crayfish) samples from this reach also show somewhat elevated concentrations.

Elevated A areas of PCDD/F contamination in sediment elsewhere in the Study Area (e.g., RM 11E, nearshore adjacent to Gunderson, Swan Island Lagoon, International Slip) are not associated with documented sources and pathways. This pattern indicates that not all point sources of PCDD/Fs to the Study Area have been identified. However these locations are coincident with areas showing elevated concentrations of other indicator chemicals, particularly PCBs and/or PAHs (see Section 10.2.1 and 10.2.4). are located between RM 6.5 and 7.5, adjacent to Arkema and Rhone-Poulanc. The relatively low density of PCDD/F data makes surface to subsurface concentration

Commented [Int8]: Value to be added following dispute resolution

gradients difficult to discern on a Study Area-wide basis. In general, however, there do not appear to be strong trends with depth.

The quantification of current external loading estimates to the Study Area was limited to upstream surface water, subsurface sediment advection, and atmospheric deposition. The highest estimated current input to the Study Area is from upstream surface water. An apparent increase in the estimated surface water flux of PCDD/Fs from upstream to downstream within the Study Area may reflect contributions from a combination of other external loading terms (e.g., stormwater, advective transport, and atmospheric deposition) or internal fate and transport processes (e.g., sediment resuspension).

In the BHHRA, total dioxin TEQ resulted in risk estimates that exceeded a cancer risk of 1×10^{-4} and/or a non-cancer HQ of 1 for both localized and Study Area-wide exposures for fish and shellfish consumption. For direct contact with in-water sediment, cancer risks exceed 1×10^{-4} for tribal fishers on a localized basis in the vicinity of RM 7W, and exceed 1×10^{-6} for in-water workers and divers (localized basis) and for high- and low-frequency fishers (localized and Study Area-wide basis). In the BERA, total TEQ was recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink populations. Total TEQ also poses risk of adverse effects to river otter, spotted sandpiper, and osprey populations and to bald eagles. These latter risks are lower than the risk to the mink population.

The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the FS fate and transport modeling. Total PCDD/F levels in surface sediments in elevated subareas located proximal to likely sources do not appear to be decreasing over time (i.e., surface and subsurface sediment concentrations are similar) despite the observation that sediment entering the Study Area appears to be comparable to upriver background levels.

10.2.3 Total DDx

The Study Area graphical CSM for DDx compounds is presented in Panels 10.2-3A–C. Loading information is summarized in Figures 10.2-7 through 10.2-9.

DDT was widely used as an insecticide from about 1943 (Porter 1962) until 1972, when it was banned for most uses in the United States, because of its toxicity to wildlife. DDE and DDD are the primary metabolites of DDT, but technical-grade DDT may also contain DDE and DDD as impurities (ATSDR 2002d). DDT was released historically to air and soil through widespread spraying of crops and forests, and for mosquito control. Releases also occurred at more local scales at pesticide manufacturing and storage facilities. Ongoing releases occur in countries where its use is not banned, and some of these releases can be transported globally through the atmosphere. Because DDT is no longer produced and sold in the United States, significant new releases to the environment are uncommon. Consequently, most of the mass of DDx found in the

Commented [Int9]: The use of DDT in the USA was not entirely banned. It has been used control plague carrying fleas in the SW USA as well as for control of insects aboard certain vessels entering US waters as well as by the US Navy for these purposes

Study Area is derived from historical sources. However, DDx compounds continue to be introduced to the LWR through a variety of environmental pathways, as described in Section 10.2.3.2.

DDT degrades slowly via abiotic and microbially-mediated processes to the more persistent DDx compounds, DDE (under aerobic conditions) and DDD (in anoxic systems; EPA 2000b). In the environment, DDx compounds are persistent and are readily bioaccumulated in aquatic organisms (EPA 2000b). While there is no clear evidence that DDT, DDD, or DDE causes cancer in humans, there is sufficient evidence of carcinogenicity of these substances in rodents, which has led to their classification as probable human carcinogens (ATSDR 2002d).

10.2.3.1 DDx Contaminant Distribution

For this discussion, elevated DDx concentrations in surface sediments are defined here as exceeding 1,000 µg/kg (the orange or red sediment polygons on the panels). Within the Study Area, the highest ~~total~~ DDx concentrations in sediments ~~a~~ are limited to localized areas in nearshore zones. The highest reported concentrations in sediment are located in the western nearshore zone between RMs 6 and 7.5, and are associated with known upland sources. Concentrations are typically greater in the subsurface than in the surface layer, indicating DDx sources are primarily historical. The concentrations of DDx in surface sediments are greater in the Study Area than those in the upriver, downtown, Multnomah Channel, and downstream reaches. Other localized areas of elevated DDx sediment concentrations are located at RM 8.8W, RM 6.5W, at the mouth of Swan Island Lagoon, at RM 4.8W (subsurface sediments only) and at the head of the International Slip. DDx has not been identified as an upland COI at sites in these areas. When DDx concentrations are averaged on a river-mile basis, the greatest difference between surface and subsurface sediment concentrations is observed in the western nearshore zone between RM 7 and 8, where the highest concentrations are found at depths greater than 10 ft (Figures 5.1-11 and 5.1-12; Panels 10.2-3A–B). —This again points to predominantly historical sources and higher past loads. Areas of high DDx concentrations are more common downstream of RM 8 (where concentrations are typically in the 5 to 100 µg/kg range) compared with upstream of this location, where concentrations are typically less than 5 µg/kg.

Commented [Int10]: Chemical-specific definition of “elevated” concentrations should be retained for clarity and consistency (EPA edits removed this text for DDx, PCBs, PAHs, BEHP, aldrin/dieldrin, metals, and TBT, but retained for PCDD/Fs and chlordanes).

DDx concentrations in surface water transect samples collected at the upstream end of and upstream of the Study Area and upriver (Panel 10.2-3A) in high-flow conditions (Figure 5.3-59) are indicative of inputs from upriver areas. The results of single-point samples collected downstream of RM 7.5, and the transect at RM 6.3 exhibit elevated DDx concentrations in both low-flow and high-flow conditions (Figure 5.3-59; Panel 10.2-3A). Results of surface water samples collected at RM 2 do not exhibit elevated concentrations although results from the surface water transect at the mouth of Multnomah Channel exhibited elevated total DDx concentration during low-flow conditions (Panel 10.2-3A; Figure 5.3-59). Total DDx concentrations in the sediment trap samples were generally low throughout most of the Study Area, although higher concentrations were measured in traps at RM 6 and 7.5W and in Swan Island Lagoon in

the summer and fall periods. The elevated DDx concentrations reported at RM 11E in the fall period are likely an artifact resulting from analytical interferences associated with PCB also detected in that sample.

Elevated total DDx concentrations in TZW samples are found near RM 7.2W; TZW samples from other areas were not analyzed for DDx precluding comparisons with other areas.

The highest concentrations of DDx in biota are found where sediments levels are highest, and most extensive near RM 7 (Figure 5.5-12a-j, Panel 10.2-3C).

~~10.2.3.1~~10.2.3.2 Potential DDx Sources and Pathways

Within Portland Harbor, DDx was historically associated with pesticide manufacturing and storage facilities. It also was released historically to air and soil through widespread spraying of crops and forests, and for mosquito control. Some riverbank facilities and other upland properties likely applied DDT for mosquito control before it was banned. Known historical chemical manufacturing and/or storage sites are presented on Map 3.2-5, and include Arkema, Rhone Poulenc, and the Shell Terminal at Willbridge. As shown on Table 10.2-4 and Panels 10.2-3A–C historical known complete or likely complete pathways for DDx have been identified at four sites. Of these, the groundwater pathway is included at two sites, stormwater at four sites, and riverbank erosion at two sites. Historical known complete or likely complete pathways for DDx were not identified for overwater releases or overland runoff. Currently known complete or likely complete pathways for DDx have been identified at three sites and include groundwater migration, stormwater, and riverbank erosion at one site. No current known complete or likely complete pathways were identified for overwater releases or overland runoff.

Areas of DDx contamination are found downstream of RM 7, and may be related, in part, to downstream transport of sediment in the relatively higher energy areas that extends from about RM 7 to RM 5 (see Section 3.1.5). DDx concentrations observed in other upstream and downstream areas are not associated with known sources (Panels 10.2-3A–B).

Historical DDx sources to the Study Area included upstream surface water, stormwater, and riverbank erosion (Table 10.2-4). The historical releases from these pathways are not quantifiable. Overall, current DDx inputs to the Study Area are much lower than historical inputs because DDT is no longer manufactured or used in widespread spraying applications and because waste management practices have greatly improved. However, elevated DDx concentrations in surface sediments and in other media, including biota, at RM 6.8 to 7.5W, and at RM 8.8W indicate localized, external inputs or ongoing internal mass transfer of historical DDx from subsurface/surface sediments to other media by processes such as sediment resuspension and biological uptake.

The most significant current influx of DDx to the Study Area is upstream surface water, while stormwater transport is the most significant current pathway for DDx to enter the Study Area from adjacent upland sites. DDx was detected in stormwater from each land use area sampled, outfalls associated with the heavy industrial land use category and selected individual non-representative outfalls contributed the majority of the estimated DDx stormwater load (see Section 6.1.2.3). Current and/or historical known or likely complete pathways for DDx in stormwater have been identified at Arkema, Rhone Poulenc, Metro Central Transfer Station, the Willbridge Terminal, and City of Portland Outfalls OF-22 and OF-22B. Most of the facilities either drain stormwater or infiltrate contaminated groundwater to shared conveyances (Saltzman Creek, OF-22B, or OF-22). Source control measures taken at the Arkema site have largely eliminated the stormwater pathway from this site. A historical likely complete pathway for DDx in stormwater has also been identified for Willbridge Terminal. No potentially complete current or historical overland transport pathways or historical overwater releases of DDx have been identified for any sites.

Commented [Int11]: Retain elements of the discussions of current and historical pathways and sources for stormwater/wastewater/overland, groundwater, overwater, riverbank, atmospheric, and upriver. This additional detail supplements the preceding paragraphs with information important to understanding sources and pathways in a large and complex site.

Known or likely historical DDx groundwater pathways have been identified at Arkema and Rhone Poulenc. Cleanup efforts conducted by Rhone Poulenc are expected to eliminate the preferential groundwater pathway to stormwater. DDx is present in upland groundwater plumes at the Arkema site, and it has been detected in nearshore wells. Groundwater controls at the Arkema site have greatly reduced the groundwater pathway.

Known or likely complete Historical DDx riverbank erosion pathways have been identified at Arkema and Willbridge Terminal. At Arkema where the riverbank area received miscellaneous fill that included miscellaneous materials from spent chlorine cells for several years (ERM 2005). The fill material included clean soil. In addition, dredge spoils were deposited on the riverbank. Riverbank erosion at the site has also been identified as a known complete current pathway for DDx.

There are no known atmospheric sources of DDx within the Study Area.

Upriver of the Study Area, DDx was identified as a COI for two sites based on hazardous substance/waste type characterizations. Of these two sites, DDx was detected in sampled media at the OHSU Moody Ave. Units A, B, C (including #1401 SW Moody Right-of-Way) property. Potential pathways to the river include groundwater and stormwater/surface runoff from the site. Upstream DDx sources with pathways potentially impacting surface water may be important since upstream surface water represents the largest current loading term for DDx to the Study Area.

~~10.2.3.2~~10.2.3.3 Loading, Fate, and Transport of DDx

Total DDx loading, fate, and transport assessment for the Study Area in a typical year is summarized on Figures 10.2-7a-b through 10.2-9. Estimates are for current conditions, and likely differed historically. Much of the total DDx mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

Historical DDx sources to the Study Area included upstream surface water, stormwater, and riverbank erosion (Table 10.2-4). The most significant current influx of DDx to the Study Area is upstream surface water, and is associated with both the dissolved and suspended particulate fraction. The majority of the mass load enters the Study Area during high-flow conditions (Figure 6.1-6), indicating that erosion of upstream agricultural areas is an important upstream source. While additional DDx mass also enters and remains in the Study Area with upstream bedload sediments, it has not been quantified. Total DDx loads in upstream surface water (RM 11) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are comparable. Additional DDx loading, in order of importance, are stormwater⁸, atmospheric deposition to the river surface (atmospheric deposition to the watershed is included in the stormwater term), groundwater advection through subsurface sediments, and upland groundwater plume discharge.

Internal DDx fate and transport for the Study Area is shown on Figure 10.2-8. DDx transport to the water column due to pore water advection through surface sediments is only process for which quantitative estimates were developed. Other internal fate and transport processes are depicted on Figure 10.2-8 on a qualitative basis only. DDx partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics.

DDx loads from permitted non-stormwater point source discharges are not expected to be significant and estimates were not generated.⁹ Estimates of DDx loading from upland soil and riverbank erosion also were not assessed in the RI due to a paucity of data for riparian soil DDx concentrations and erosion rates.

Total DDx load in surface water increases moving downstream through the Study Area (Figure 10.2-9), with the largest DDx stormwater inputs entering between RM 6.8 and 7.4. Upland groundwater plume loads are highest between RM 7 and 8. Estimated

⁸ The DDx stormwater loading term developed in the RI is dominated by the estimated load from a single “non-representative” site. As discussed in Section 6, the estimated loads from this non-representative site are subject to a high degree of uncertainty related to extrapolation of stormwater concentrations measured in the non-representative outfalls (<10 percent of the drainage sub-basin) to areas that are larger than the catchments that drain to the non-representative outfalls.

⁹ NPDES permitted wastewater discharge is not expected to contain significant amounts of total DDx relative to other loading terms.

current DDx annual loads from advection through subsurface and surface sediments follow the patterns of DDx sediment concentrations (Figure 6.1-59), with the highest loads between RM 7 and 8.

10.2.3.4 DDx CSM Summary

DDx compounds are widely detected at relatively low levels in Portland Harbor surface and subsurface sediments; other abiotic media, such as surface water, sediment traps, and TZW; and fish and shellfish tissues. Total DDx concentrations in sediments in much of the channel and in most of the Study Area upstream of RM 8 are generally less than 5 µg/kg, downstream of RM 8, concentrations in the channel are typically between 5 and 10 µg/kg. This compares to an upriver background level of XX µg/kg (on OC-equivalent basis).

Commented [Int12]: Value to be added following dispute resolution

Elevated DDx levels are found in several nearshore areas. The highest concentrations are associated with known DDx historical sources (Arkema, Rhone Poulenc, and Willbridge Terminal) are found in surface, and especially subsurface sediments along the western shore of the river from about RM 6.8 to 7.5. Concentrations of DDx compounds in surface sediments throughout the Study Area are generally lower than subsurface sediment concentrations, indicating greater inputs or releases historically that have been reduced or eliminated over time.

Relatively elevated DDx levels are also seen in the surface water, sediment traps, TZW, and biota samples from the RM 6.8 to 7.5 area. Sediment concentration gradients along this western nearshore area and channel edge downstream from this source area suggest the downstream transport of DDx in sediments and possible inputs from other point sources. This pattern is consistent with the hydrodynamic and sediment transport characteristics in this portion of the Study Area, particularly from RM 5 to 7. Less extensive areas with elevated DDx occur at Gunderson, at the head of International Slip, RM 4.8W (subsurface only), and at the Cascade General Shipyard at depth in the sediment column.

Other areas of elevated total DDx sediment concentrations are smaller in extent and are located at RM 8.8W, RM 6.5W, at the mouth of Swan Island Lagoon, and in subsurface sediments only at RM 4.8W and the head of the International Slip. There are no known or potential sources of DDx in these areas and this may represent upland source data gaps. Elevated subsurface concentrations, e.g., in International Slip, likely reflect historical sources.

Estimates of quantifiable external loading terms indicate that the highest current external inputs to the Study Area are from upstream surface water. Estimates of DDx fluxes in surface water at RM 2, the downstream end of the Study Area, and in Multnomah Channel suggest that slightly more DDx mass may be leaving the Study Area downstream in surface water than entering the Study Area from all quantified sources. This possible increase may reflect an internal mass transfer of DDx from bedded sediment to the water column, likely through sediment resuspension. The

relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted at the Site for the FS.

The baseline risk assessments reflect the observed distribution of DDx compounds in the Study Area with the highest risks associated with the RM 7W area. In the BHHRA, DDD, DDE, and DDT were identified as contaminants potentially posing unacceptable risks because of estimated cancer risk exceedances of 1×10^{-6} and HQs greater than 1 from the ingestion of fish. The BERA concludes that DDx compounds in sediment pose potentially unacceptable risk to the benthic community only on the western side of the river between approximately RM 6.8 and 7.4.

10.2.4 Total PAHs

~~PAHs are present at a wide range of concentrations throughout the Study Area in all media.~~ The graphical CSM for PAHs is shown on Panels 10.2-4A–C. PAHs are a large chemical group composed of more than 100 chemicals that are constituents of crude and refined oil, shale oil, coal tar, and creosote. PAHs are also formed during the incomplete combustion of organic materials including coal, oil, gas, wood (wood stoves, fireplaces), garbage (municipal waste incineration), or other organic substances, such as tobacco. Natural sources of PAHs include volcanoes and forest fires. The largest historical sources of the high concentrations of PAHs in Study Area sediments were localized releases, such as historical industrial direct discharges, from facilities along the Study Area that formerly or currently use, manufacture, and/or store products and waste materials containing PAHs (petroleum oils and coal-based products, petroleum product use and storage, used oil, and asphalt roofing materials).

PAHs may accumulate in benthic organisms, fish, and other organisms that ingest sediments while feeding. However, biomagnification through the food chain is expected to be minimal as fish, mammals, and birds all possess the ability to metabolize PAH compounds.

10.2.4.1 PAH Contaminant Distribution

PAHs are present at a wide range of concentrations throughout the Study Area in all media. For this discussion, concentrations greater than 10,000 µg/kg in sediment, 50,000 pg/L in surface water, 100 µg/L in TZW, and 100 µg/kg in biota are considered elevated (the orange or red sediment symbols and polygons on Panels 10.2.4A–C).

On a harbor-wide basis, elevated PAH concentrations in sediments generally occur downstream of RM 7 in nearshore areas proximal to local upland sources (Maps 5.1-9 and 5.1-10a–m; Panels 10.2-4A–B) offshore of Siltronic, Gasco, Marine Finance, and Foss Brix. Elevated PAH concentrations are also observed in surface and subsurface sediments in the navigation channel between RM 4 and 6.5. Other areas of elevated total PAH concentrations in surface sediments include Mar Com South (RM 5.5–5.6E), Terminal 4 Slip 3 and Wheeler Bay (RM 4.3–4.6E), Slip 1 (RM 4.3E), and the International Slip (RM 3.7–3.8E).

Total PAH concentrations are generally higher in subsurface sediments within the Study Area as a whole (Panel 10.2-4A-B; Maps 5.1-10a-m; Figure 5.1-45), pointing to higher historical inputs to the Study Area. The most notable exception to this pattern is the navigation channel at RM 5 to 6.5 where the PAH concentrations in surface sediment are greater (Maps 5.1-10f-g), indicating suggesting possible downstream transport of PAHs in surface sediments once they reach the channel from adjacent or upstream nearshore zones. This is consistent with the relative dynamic channel environment found from RM 5 to 7 (see Section 3.1.5). Other exceptions to the general pattern of higher subsurface total PAHs include Swan Island Lagoon and Multnomah Channel, where higher PAH concentrations are observed in the upper portion of the sediment column and not just the surface layer.

PAH compositions in sediment trap and high-flow surface water particulate samples were generally similar to that of the sediments, indicative of increased bedded sediment resuspension during higher flow periods. PAH composition in sediment trap and surface water during low-flow and stormwater-influenced surface water particulate samples generally did not correlate well, indicating that current, lateral and upstream sources differ in composition from the PAHs in the bedded sediments. Total PAH concentrations in surface water are elevated predominately during low-flow conditions (Figure 5.3-77), further illustrating the localized nature of the PAH sources in the Study Area. PAH concentrations in sediment trap samples are notably elevated at RM 6W in all seasons sampled (Figure 5.2-19). Less distinct spikes were measured from May to August in Swan Island Lagoon, and from August to November in Multnomah Channel.

Total PAH concentrations in TZW were reported in areas that correspond with elevated areas of sediment concentrations. The highest concentrations reported in biota samples also correspond with areas where PAH concentrations sediment are elevated.

10.2.4.110.2.4.2 Potential PAH Sources and Pathways

PAHs are associated with bulk fuel storage, MGP, ship and automobile scrapping, asphalt roof manufacturing, use of lubricants, hydraulic, and fuel oils in a large number of industrial and commercial operations, and leaks from vehicles and machinery.

Known complete or likely complete historical pathways for PAHs have been identified at 56 sites, and include stormwater (48 sites), groundwater (11 sites), overwater releases (21 sites), overland transport (10 sites), and riverbank erosion (14 sites). Current known complete or likely complete pathways for PAHs have been identified at 30 sites and include stormwater (16 sites), groundwater (9 sites), overwater releases (14 sites), overland transport (3 sites), and riverbank erosion (4 sites). This information is summarized in Table 10.2-5 and on Panels 10.2-4A-C. Known or likely current and historical upland sources that correlate with areas of high concentrations in sediment include the former MGP site at Gasco, dry dock operations at Cascade General in Swan Island Lagoon, the International Slip, MarCom, bulk fuel terminals at ARCO, ExxonMobil, and Kinder Morgan, and historical releases from McCormick and Baxter

in Willamette Cove. Based on reported PAH concentrations in sediment near outfalls draining facilities, it is also likely that stormwater/wastewater/overland transport releases occurred concurrently at Burgard Industrial Park (WR-123), Siltronic and Gasco (OF-22C), and Greenway Recycling, PGE-Forest Park, Willbridge Terminal, Front Avenue LP, and Chevron Asphalt (OF-19). Because wastewater discharges in CSO areas are regulated through municipal pretreatment permits, PAHs are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

Current known and likely complete pathways for migration of PAHs in groundwater have been identified at nine facilities, most of which are associated with bulk fuel storage (Table 10.2-5). Overwater releases are likely to have occurred in association with overwater operations such as fuel transfers and spills, drydock and berth operations, overwater maintenance operations, vessel servicing and emissions, ship repair and maintenance activities, direct discharges of PAH contaminants (tar, oil) to the river, releases during product loading/unloading at docks, and tug and barge operations.

Known complete groundwater pathways were identified at the Siltronic and Gasco sites. Potentially complete groundwater migration pathways were identified at Willbridge Terminal, Kinder Morgan Linnton, ARCO, and ExxonMobil. However, results from TZW samples collected offshore of these four sites indicate that the role, if any, of groundwater transport of PAHs is minor and is not significantly influencing TZW and sediment chemistry.

Groundwater discharge is likely to have been a more important source historically and known or likely complete historical pathways have been identified at eight sites.

Bank erosion likely played a bigger role historically as well, particularly during construction in places where contaminated sediments or manufacturing material and debris from upland activities were used as fill. Based on limited riverbank sampling, riverbank erosion is a historical known or likely complete pathway for PAHs at 13 sites: Crawford, Gasco, Gunderson, MarCom South Parcel, Marine Finance, Port of Portland Terminal 4, Slip 1 and Slip 3, Premier Edible Oils, Siltronic, Sulzer Bingham, Triangle Park, Willamette Cove, and the Willbridge Terminal facility.

Table 4.5-1 lists ECSI sites upstream of the Study Area (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from DEQ ECSI database. Environmental investigation or cleanup of sediments or upland areas have been ordered for some of these properties, suggesting that they could be now, or in the past have been, significant sources of PAHs to the Willamette River. PAHs were identified as COIs for 22 sites based on their hazardous substances/waste types, and PAHs were detected in sampled media at 14 sites.

~~10.2.4.2~~10.2.4.3 Loading, Fate, and Transport of PAHs

Total PAH loading, fate, and transport in the Study Area in a typical year are summarized on Figures 10.2-10a, 10.2-11a, and 10.2-12, relative loads for LPAHs and HPAHs are presented on Figures 10.2-11b and 10.2-11c, respectively. While these loading estimates are for current conditions, much of the total PAH mass in the Study Area is attributable to historical loading.

Advection through subsurface sediments, upstream surface water, and upland groundwater plumes are estimated to contribute comparable (within an order of magnitude) total PAH loads to the Study Area (Figures 10.2-10a and 10.2-11a). Estimated external PAH loads associated with stormwater, atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources are one to two orders of magnitude lower than the other external terms. LPAHs contribute the bulk of the total PAH load for all loading terms evaluated quantitatively (Figures 10.2-11b and 10.2-11c). The majority of the total PAH mass load from upstream surface water enters the Study Area during high-flow conditions (Figure 6.1-8). Upstream surface water total PAH and LPAH loads are associated primarily with the dissolved fraction, whereas HPAH loads are slightly higher in the particulate than in the dissolved fraction. PAH load in surface water increases moving downstream through the Study Area, particularly downstream of RM 7, in both the particulate and dissolved fractions (Figure 10.2-12) under all flow conditions. PAH loading terms that were assessed qualitatively are also shown on Figures 10.2-11a–c. PAH volatilization from the water column may be significant for LPAHs, although this was not evaluated.

Fate and transport processes internal to the Study Area for total PAHs, LPAHs, and HPAHs are also shown on Figures 10.2-11a, 10.2-11b, and 10.2-11c, respectively. Quantitative estimates were developed only for transport to the water column due to pore water advection through surface sediments, and this term is lower in magnitude than the subsurface advective loading term for total PAHs and LPAHs, and similar in magnitude for HPAHs. Estimates of PAH loading from upland soil and riverbank erosion were not assessed due to a paucity of data for riparian soil PAH concentrations and erosion rates.

The data shown on Figure 10.2-12 suggest that the total PAH load in surface water increases moving downstream through the Study Area,¹⁰ particularly downstream of RM 7. Most of the load from the two largest external lateral loading terms, subsurface sediment advection and upland groundwater plumes, enters the Study Area between RM 6 and 7.

¹⁰ At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3).

10.2.4.310.2.4.4 Relationship of Sources to Distribution of PAHs CSM Summary

The concentration of total PAHs in Portland Harbor sediments varies from levels that are likely within the range of urban background to very high concentrations in localized areas associated with historical sources. The portion of the Study Area upstream of RM 7 (not including Swan Island Lagoon) is characterized by widespread PAH concentrations less than 500 µg/kg. In contrast, downstream of RM 7 and away from the high concentration areas associated with known sources, and also in Swan Island Lagoon, PAHs concentrations are generally 1,000–5,000 µg/kg. For comparison, the upriver background PAHs level is XX µg/kg (on OC-equivalent basis).

Commented [Int13]: Value to be added following dispute resolution

PAH contamination in sediments is associated with known or likely historical and current sources. Two areas of elevated PAH concentrations, offshore of the Gasco former MGP site at RM 6.5W and the Port's Terminal 4, Slip 3, are the focus of early cleanup actions independent of the harborwide RI/FS. Releases associated with Gasco and other known and potential sources along the west side of the river between RM 6 and 7 have resulted in elevated nearshore PAH concentrations in both surface and subsurface sediments. Downstream concentration gradients both nearshore and in the navigation channel in this relatively high-energy portion of the river suggest downstream transport of PAHs extending approximately to the Multnomah Channel entrance. In the navigation channel from RM 5 to 6, high surface sediment concentrations relative to subsurface levels also suggest transport of material through this reach rather than long-term accumulation. Other, less extensive areas of PAH sediment contamination in the Study Area, are associated with known or likely sources including offshore of Cascade General, in Willamette Cove (subsurface sediments), offshore of Mar Com, at the head of the International Slip, and at several nearshore locations along the west bank from RM 3 to 5.

In general, subsurface PAH levels exceed surface levels, indicating greater inputs or releases historically that have been reduced or eliminated over time. Elevated PAH levels in other media—TZW, surface water, sediment traps, and biota (particularly clams)—are largely restricted to the most extensive area of elevated sediment concentrations, along the western shore around RM 6.

Overall, current PAH inputs to the Study Area are much lower than historical inputs because material handling and waste management practices have greatly improved. Measured elevated levels of PAHs in surface sediments and other media including surface water (e.g., Swan Island Lagoon, RM 7.4W, RM 6.8W) and biota in specific areas (e.g., International Slip; Terminal 4, Slips 1 and 3; offshore of Gasco; and Swan Island Lagoon) indicate that localized inputs and/or internal mass transfer of sediment PAHs by processes such as sediment resuspension likely affect other media. Potentially important current pathways from upland sources include groundwater plumes, overwater releases, overland transport, and riverbank erosion.

Empirical estimates of current external PAH loads (mass/yr) to the Study Area indicate that advection through subsurface sediments, upstream surface water, and upland groundwater plumes contribute comparable total PAH loads to the Study Area, whereas loads associated with stormwater, direct atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources are considerably less important. These data also suggest that surface water PAH mass loads increase from upstream to downstream, likely reflecting inputs from the other external loading terms, which peak at RM 6 to 6.9. Internal mass transfer from bedded surface sediments to surface water/biota from sediment resuspension erosion was not quantified but likely also contributes to this pattern.

PAHs were identified as contaminants potentially posing unacceptable risks for the BHHRA based on fish and shellfish consumption and direct contact exposures. Human health and ecological risks from PAHs are geographically limited in extent, reflecting the limited areas of very high sediment levels, and are concentrated in the RM 4 to 6W area. In the BERA, PAHs are strongly associated with the benthic risk area from RM 5.1 to 6.9 on the west side of the river, where concentrations are elevated in sediment and TZW. Negligible risks to receptors other than the benthic community are expected based on the low magnitude and frequency of exceedances in other areas of the river.

10.2.5 Bis(2-ethylhexyl)phthalate

A graphical CSM for BEHP in the Study Area is presented on Panels 10.2-5A–C. Phthalates are manufactured, colorless liquids with little or no odor. The primary sources of phthalate emissions are the industries that manufacture it or use it in productions, such as the chemical industry, the plastics industry, the cosmetic industry, machinery manufacturers, and manufacturers of plywood and millwork. Phthalates are commonly added to plastics and paint to make the finished product more flexible (ATSDR 2002c) and are widely used as plasticizers in PVC resin. In addition, phthalates are common components of detergents and carriers in pesticide formulations (Xie et al. 2005). Releases to the environment can occur as direct spills from industrial facilities that manufacture or use these chemicals. More commonly, releases occur by leaching of low volumes of phthalates from the wide variety of products that contain them (ATSDR 2002c). Despite its low vapor pressure, BEHP is ubiquitous in the atmosphere due to its widespread use in plastics. BEHP is present in the atmosphere in both the vapor phase and associated with particulates, is subject to both wet (rain and snow) and dry (wind and settling) deposition on the earth's surface (ATSDR 2002c).

The behavior of BEHP in the Study Area environment is largely defined by its high hydrophobicity (the central estimate of log K_{oc} is 7.4). Due to this hydrophobicity, BEHP has a strong tendency to sorb to solids and organic matter in surface water and in sediment-pore water environments. As such, it is not expected to migrate significantly in groundwater. Because of its low vapor pressure, volatilization is a minor loss

mechanism for BEHP, particularly when sorbed to solids. BEHP is subject to fairly rapid degradation in the atmosphere, but much slower abiotic and microbially-mediated degradation processes under aerobic conditions in sediment and surface water (HSDB 2006; ATSDR 2001). While its physical properties indicate that BEHP would be expected to bioconcentrate in aquatic organisms; however, Study Area biota results suggest that BEHP is readily metabolized.

10.2.5.1 BEHP Contaminant Distribution

For BEHP, concentrations greater than 3,000 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 0.8 pg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-5A–C).

Elevated BEHP concentrations in sediments were reported, with minor exceptions, in nearshore areas outside the navigation channel and proximal to local upland sources (Maps 5.1-12a–m and Panels 10.2-5A–B), and are observed in Swan Island Lagoon and in the International Slip (RM 3.7–3.8E), and along the riverside of Schnitzer/Calbag site RM 3.8–4.1E, RM 7.6E, RM 9.7W, RM 8.8W, RM 8.3W, RM 7.6W, , and offshore of RM 7.1, and RM 10 in the navigation channel. Elevated subsurface concentrations were less widespread, and typically observed in areas with elevated surface concentrations. Exceptions were noted at RM 10.5W and at RM 5.7W.

Only one surface water sample collected during high-flow conditions at RM 8.6W showed elevated concentrations of BEHP. BEHP concentrations in sediment trap samples did not vary widely spatially or temporally throughout and upstream of the Study Area, with the exception of notably elevated levels measured in the only two samples (summer, fall) collected in Swan Island Lagoon (see Figure 5.2-22).

Because of BEHP's hydrophobic nature, groundwater is unlikely to be a significant historical or current pathway for BEHP migration into the Study Area and was not included in the TZW sampling program. Thus, no data exist to corroborate this hypothesis.

BEHP was detected in laboratory-exposed clams and worms, mussels, and fish. However, it was not detected in crayfish, juvenile Chinook, or carp. The highest reported concentrations in biota were generally detected on both sides of the river near RM 4 and above RM 9.5, and one location at the downstream end of Swan Island. With the exception of the surface sediment on the east bank near RM 4 and subsurface sediment at the downstream end of Swan Island, elevated BEHP concentrations in biota do not correlate well with elevated concentrations in sediment.

10.2.5.110.2.5.2 Potential BEHP Sources and Pathways

Historical known complete or likely complete pathways include stormwater (28 sites), overwater releases (three sites), overland runoff (one site), riverbank erosion (two sites), and groundwater (one site) (Table 10.2-6). Current known complete or likely complete

pathways for BEHP have been identified at 16 sites and include groundwater (three sites), stormwater (12 sites), and overwater releases (two sites).

BEHP is elevated in surface sediment in current and former shipyards such as Swan Island and International Slip. BEHP is likely to have been released to Swan Island Lagoon and the Portland Shipyard for many years, and continued inputs may occur from known sources and contributions from the numerous outfalls in this area. The lack of elevated levels at depth in the sediment column in the lagoon suggests low burial rates, surface sediment mixing, and/or higher recent inputs. The subsurface maximum levels near the shipyard docks indicate historically high levels and burial over time.

There are no known sources of BEHP associated with the ECSI sites that discharge stormwater to the International Slip. However, this area has an auto shredding facility and BEHP has been identified with metal scrapping (see Section 3.2.1.4). It was also the former location of a large shipyard owned by the Oregon Shipbuilding Corporation. Several metals facilities (either fabrication or scrapping) and a historical paint spill area have been identified as BEHP sources that discharge to OF-19 at RM 8.3W. Identified sources of BEHP at RM 8.8W include Gunderson (outfalls from the Marine Barge Paint and Blast Area) and multiple facilities discharging to OF-18.

BEHP migration by overland transport has been identified at the MarCom South Parcel as a historical likely complete pathway. Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall. Except for Cascade General and Mar Com South, no current known or likely complete BEHP overwater pathways have been identified. Groundwater Discharge been identified as a current or historical complete pathway at Premier Edible Oils, Triangle Park, and the Willbridge Terminal facility. There are no known atmospheric sources of BEHP within the Study Area.

No upriver watershed sources of BEHP have been identified.

10.2.5.210.2.5.3 Loading, Fate, and Transport of BEHP

BEHP loading, fate, and transport assessment for the Study Area in a typical year is presented on Figures 10.2-13a through 10.2-15. The highest relative current external inputs are from upriver surface water,¹¹ with relatively minor additional contributions from Study Area stormwater and advection through subsurface sediments. The total BEHP loads in surface water upstream and the downstream boundary of the Study Area are generally comparable. Although the current upstream surface water load estimate exceeds the other loading terms, there is no indication that the surface water load is responsible for spatial distribution of the BEHP observed in Study Area sediments, and much of the surface water load appears to pass through the Site. As noted, the off-

¹¹ The surface water data set only has total concentrations for BEHP; therefore, Figure 10.2-13a presents only the total estimated surface water loading rates for this chemical.

channel and nearshore elevated BEHP concentrations appear to be associated with localized upland sources and pathways.

A cross-media comparison and statistical assessment of surface sediment, sediment trap samples, and particulate suspended solids on a Study Area-wide basis (Tables 10.2-2a-b and Figure 10.2-13b) show that the mean BEHP concentrations differ significantly across all media, with the surface sediment showing the highest average concentrations, followed by surface water particulates and then sediment trap samples.

~~10.2.5.3~~ 10.2.5.4 Relationship of Sources to Distribution of BEHP CSM Summary

The larger areas of elevated BEHP concentrations are located in off-channel areas (Swan Island Lagoon, International Slip, and the riverside along Schnitzer/Calbag). Sediment trap samples were elevated in Swan Island Lagoon. Other areas of elevated sediment concentrations are limited to a few, typically small, widely scattered nearshore areas. With the exception of sediments near the Cascade General shipyard, there are generally more elevated samples in the surface than subsurface sediments, suggesting recent or current sources exist and that historical inputs were not generally elevated relative to current levels. Clams, crayfish, and sculpin tissue BEHP levels are generally not elevated in the Study Area; smallmouth bass samples were elevated at selected areas, including RM 9.5 to 11.5 and near RM 4.

Within the Study Area, several historical and current sources contributed BEHP to the river primarily through the stormwater pathway. Elevated BEHP concentrations in sediment are primarily found in Swan Island Lagoon, where ship construction, repair, painting, and maintenance operations have been ongoing since the 1940s, as well as other manufacturing sites with stormwater drainage to the lagoon. BEHP concentrations also correspond to locations in the harbor where there are equipment manufacturing, marine salvage, and scrap metal recycling operations.

The loading calculations indicate that the highest relative current external inputs are from upriver surface water, with relatively minor additional contributions from Study Area stormwater and advection through subsurface sediments.

The BHHRA found that BEHP resulted in cancer risk estimates greater than 1×10^{-4} and HQs greater than 1 for consumption of whole-body smallmouth bass at RM 4. BEHP resulted in cancer risk estimates greater than 1×10^{-6} for consumption of whole-body and fillet smallmouth bass and whole-body brown bullhead. In the BERA, BEHP was identified as posing potentially unacceptable risk to benthic invertebrates, fish, amphibians, and aquatic plants based on the surface water LOE; and to benthic invertebrates and fish based on the tissue residue LOE. Negligible risks are expected based on the low magnitude and frequency of exceedances.

10.2.6 Total Chlordanes

A graphical CSM for total chlordanes Study Area is presented on Panels 10.2-6A–C. Chlordane is a manufactured chemical that was used as a pesticide on crops, including corn and citrus, and on home lawns and gardens in the U.S. from 1948 to 1988. It was also used from the 1950s to the 1980s to prevent or eliminate termites. Some of its trade names are Octachlor and Velsicol 1068. Chlordane does not occur naturally in the environment. The EPA banned all uses of chlordane, with the exception of termite control, in 1983; all uses were banned in 1988 (ATSDR 1995). Chlordanes are hydrophobic, and to sorb to solids and organic matter in surface water and sediment. They are persistent in sediments, and subject to very slow abiotic degradation processes. The dissolved fraction in surface water is subject to volatilization. Chlordanes can bioaccumulate in the tissues of fish, birds, and mammals (ATSDR 1995).

10.2.6.1 Total Chlordanes Contaminant Distribution

Total chlordanes are the sum of oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, and cis-nonachlor. Each of these chemicals is analyzed individually and the concentrations are summed to obtain the total chlordanes value. For total chlordanes, concentrations greater than 10 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 60 pg/L in surface water samples are considered elevated (the red and orange symbols and polygons in Panels 10.2-6A–C).

Several sediment, surface water, and biota samples were reported with high detection limits for total chlordanes, some of which meet or exceed the above definition of elevated (Panels 10.2-6A–C). These high detection limits are the result of chromatographic interferences, laboratory blank contamination, mass spectrometer details related to identification of the components of total chlordanes, or limited sample sizes. High detection limits may obscure the presence of total chlordanes at a concentration below the elevated detection limit, but a high detection limit does not imply the presence of the chemical.

~~10.2.6.1.1 Sediment~~

On a harbor-wide basis, the highest detected concentrations chlordane in sediments are restricted to small, widely scattered nearshore or off-channel areas, proximal to local upland sources (Maps 5.1-13 and 5.1-14a–m and Panels 10.2-6A–B). Areas where surface and subsurface sediment concentrations are greater than 10 µg/kg include RM 5.8W-9W, and approximately RM 3E, 4E, 5.5E, and 11E. Total chlordanes were detected in a majority of surface water samples. Elevated surface and subsurface concentrations are also found in Swan Island Lagoon, RM 5.6, and International Slip along the eastern nearshore. Reported concentrations in sediment trap samples were typically low (<4 µg/kg), with no strong temporal or spatial patterns in the measured concentrations (Figure 5.2-23). Chlordanes were detected at low concentrations with varying frequency in all fish and invertebrate samples.

10.2.6.2 Potential Sources and Pathways of Total Chlordanes

The known sources of total chlordanes to the Study Area are summarized in Table 10.2-7 and on Panels 10.2-6A–C. The former Rhone Poulenc pesticide manufacturing facility is the only source of chlordanes currently identified that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9, and site groundwater infiltrates to City of Portland outfall OF-22B. However, the distribution of chlordane in nearshore sediments is indicative that other sources may also be present.

Historical known pathways for stormwater exist at the Rhone Poulenc facility, including infiltration of contaminated groundwater into the storm system and the City of Portland outfalls, specifically OF-22B (RM 6.9W). Total chlordanes have been detected in upland soils at the former Rhone Poulenc property, and stormwater is a likely complete historical pathway. Historically, manufacturing wastes from Rhone Poulenc ~~were~~ was discharged to Doane Lake, which occasionally discharged to the river via a historical drainage ditch from 1972 to 1980. This ditch entered the river near RM 6.9, an area with elevated surface and subsurface sediment total chlordanes concentrations. Currently, site stormwater is collected, treated, and discharged through WR-6, also located at RM 6.9W. No current known or likely complete overland transport pathways for total chlordanes have been identified.

Total chlordanes were detected in sediment samples collected during the PGE Willamette River Sediment Investigation, and chlordanes were likely used as a pesticide in the agricultural areas of the Willamette River watershed up until the late 1980s so upriver sources continue to exist.

10.2.6.3 Loading, Fate, and Transport of Total Chlordanes

The loading, fate, and transport of total chlordanes in the Study Area in a typical year is summarized on Figures 10.2-16a through 10.2-18. Current external inputs are dominated by upriver surface water, with relatively minor additional contributions from Study Area stormwater, advection through subsurface sediments, and atmospheric deposition to the river surface. The total chlordane upstream surface water loads are comparable with the combined loads at RM 2 and the Multnomah Channel entrance in both the dissolved and particulate fractions. Although the current upstream surface water load estimate exceeds the other current loading terms, there is no indication that this load is responsible for distribution of total chlordanes in Study Area sediments.

Cross-media comparisons of surface sediment, sediment traps, and suspended solids in surface water (Tables 10.2-2a-b and Figure 10.2-16b) show that the overall surface sediment concentrations are greater than those in surface water particulate and sediment trap samples. Surface water particulate and sediment trap concentrations are similar.

10.2.6.4 Relationship of Sources to Distribution of Total Chlordanes CSM Summary

Chlordanes have been banned for production and use since the 1980s; however, they are persistent compounds in the atmosphere due to widespread use as a pesticide. These compounds have been detected in surface and subsurface sediment, sediment traps, surface water, and fish and invertebrate samples from the Study Area. Total chlordane concentrations across most of the Study Area are comparable to upriver background levels except for a few nearshore areas that are both more quiescent and proximal to known historical sources. The largest areas of elevated total chlordanes include RM 8.8, 6.9–7.5, and 6.1–6.7 along the western shoreline, and Swan Island Lagoon and RM 3.7–3.8 (International Slip) along the eastern nearshore. Subsurface concentrations tend to be higher than surface concentrations, suggesting that historical inputs were larger than current inputs.

Within the Study Area, the only source of chlordanes currently identified is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9. Total chlordanes have been detected in upland soils and groundwater at this site, and stormwater is a complete historical pathway. Infiltration of contaminated groundwater to the storm system draining OF-22B is addressed in Section 10.2.3.2.2. However, based on the distribution of chlordanes in nearshore sediments, other sources must be present.

The loading, fate, and transport assessment for total chlordanes in the Study Area in a typical year is summarized in Figures 10.2-16a through 10.2-18. Current external inputs are dominated by upriver surface water, with relatively minor additional contributions from Study Area stormwater, advection through subsurface sediments, and atmospheric deposition to the river surface.

In the BHHRA, total chlordanes was identified as a contaminant potentially posing unacceptable risks only for the fish consumption scenario based on cancer risk estimates above 1×10^{-6} but did not result in cancer risk estimates greater than 1×10^{-4} . The BERA did not identify chlordanes as a contaminant potentially posing unacceptable risks.

10.2.7 Aldrin and Dieldrin

Graphical Study Area CSMs for aldrin and dieldrin are presented on Panels 10.2-7A–C and 10.2-8A–C, respectively. Aldrin and dieldrin are organochlorine insecticides with similar chemical structures, manufactured for agricultural use from the 1950s to 1987. From the 1950s until 1970, aldrin and dieldrin were widely used insecticides for crops and livestock. Peak production occurred in the mid-1960s. Aldrin/dieldrin ranked second—after DDT—among agricultural insecticides used in the U.S. in the 1960s (Jorgenson 2001). EPA banned the use of aldrin and dieldrin in 1974, except to control termites. By 1987, EPA banned all uses (ATSDR 2002b).

Like the other pesticides on the CSM chemical list, aldrin and dieldrin are highly hydrophobic and have a strong tendency to sorb to solids and organic matter in surface water and sediment. Aldrin is subject to abiotic and microbially-mediated degradation processes as well as photolysis in surface water. Dieldrin is more recalcitrant, subject to very slow degradation processes in the environment. Both chemicals are subject to fairly slow volatilization (ATSDR 2002b). Plants can absorb both aldrin and dieldrin from the soil. Dieldrin can bioaccumulate, while Aldrin does not bioaccumulate it readily degrades to dieldrin in plants and animals (ATSDR 2002b).

10.2.7.1 Aldrin and Dieldrin Contaminant Distribution

For aldrin and dieldrin, concentrations greater than 10 µg/kg in sediment and biota samples are considered elevated. For surface water samples, aldrin concentrations greater than 8 pg/L and dieldrin concentrations greater than 100 pg/L are considered elevated (the red and orange symbols and polygons on Panels 10.2-7A-C, 10.2-8A-C).

Several sediment, surface water, and biota samples were reported with high detection limits for aldrin or dieldrin, and some of these detection limits are at a level that meets the above definition of elevated (Panels 10.2-7A-C, 10.2-8A-C). These high detection limits are the result of chromatographic interferences, laboratory blank contamination, mass spectrometer criteria issues, or limited sample sizes. High detection limits may obscure the presence of aldrin or dieldrin at a concentration below the detection limit, but a high detection limit does not imply the presence of the chemical.

Aldrin and dieldrin contamination in sediment is generally restricted to small, widely scattered nearshore areas. Elevated concentrations were generally not detected in off-channel or navigation channel areas, with the exceptions of detections of dieldrin at RM 11.4E (which includes several samples nearshore and offshore with elevated concentrations) and the head of the International Slip (RM 3.7E). Surface and subsurface sediments results show elevated concentrations of aldrin and dieldrin at RM 6.8 to 7.5W and RM 8.8W. Overall, aldrin concentrations slightly higher in subsurface sediments, while dieldrin concentrations are generally higher in the surface sediments. Areas with high concentrations of aldrin and dieldrin are generally co-located.

Aldrin and dieldrin were detected both in particulate and dissolved surface water samples, with the dissolved fraction slightly to largely predominating with a major exception at RM 6.9. Aldrin and dieldrin were infrequently detected in sediment trap, most frequently downstream of RM 8. Tissue concentrations of aldrin and dieldrin concentrations were generally less than 10 µg/kg, although higher dieldrin concentrations were observed in sculpin collected from RM 2E-RM 4E.

10.2.7.2 Potential Aldrin/Dieldrin Sources and Pathways

The known sources of aldrin are summarized in Tables 10.2-8 and 10.2-9 for aldrin and dieldrin, respectively. ~~The CSMs are presented for aldrin on Panels 10.2-7A-C and~~

~~10.2-8A-C for dieldrin.~~ The only currently identified source of aldrin and dieldrin within the Study Area is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9. Aldrin and dieldrin have been detected in upland soils at this site, and stormwater is a known complete current and historical pathway. However, based on the distribution of aldrin and dieldrin in nearshore sediments, other sources must be present.

Historically, manufacturing wastes from Rhone Poulenc were routed to Doane Lake, which occasionally discharged to the river via a drainage ditch ~~that discharged to the river~~ near RM 6.9 ~~from 1972 to 1980~~. Currently site stormwater is collected, treated, and discharged through WR-6, also located at RM 6.9W. No current known or likely complete overwater or riverbank erosion pathways for aldrin or dieldrin have been identified. Atmospheric deposition is a potential historical and current pathway, but has not been quantified.

Jorgenson (2001) lists one facility in the Portland Harbor area, Van Waters & Rogers, as a producer and/or distributor of aldrin and dieldrin. Van Waters & Rogers, an ECSI site, was listed as being an active formulator in 1969. No information is available about releases to the environment from this facility.

10.2.7.3 Loading, Fate, and Transport of Aldrin and Dieldrin

The loading, fate, and transport assessment of aldrin in the Study Area for a typical year is summarized on Figures 10.2-19a through 10.2-21, and on Figures 10.2-22a through 10.2-24 for dieldrin. Upriver surface water is the largest estimated external loading term, dieldrin surface water loading is approximately 50 times than that of aldrin. Stormwater is the second highest estimated loading term, followed by atmospheric deposition to the river surface and advection through subsurface sediments. There is an apparent increase in aldrin loads in surface water between the upstream and downstream boundaries of the Study Area. Conversely, there is little apparent change in surface water dieldrin loads between upstream and downstream boundaries. Although the current upstream surface water load estimate exceeds the other loading terms, there is no indication that the surface water load is responsible for the spatial distribution of aldrin and dieldrin in Study Area sediments, and nearshore aldrin and dieldrin concentrations appear to be associated with localized upland sources and pathways.

Cross-media comparisons of surface sediments, sediment trap samples, and suspended solids in surface water for aldrin and dieldrin are provided in Tables 10.2-2a-b (summary statistics and statistical comparisons) and Figures 10.2-19b and 10.2-22b (box-whisker distribution plots). The OC-normalized aldrin surface sediment and sediment trap concentrations are not significantly different; concentrations in both of these media are greater than the concentration in surface water particles on a Study Area-wide basis. Dieldrin Study Area-wide surface sediment concentrations are significantly greater than the concentrations in surface water particulates, but there are

no statistically significant differences between sediment trap concentrations and sediments or surface water particulates.

10.2.7.4 Relationship of Sources to Distribution of Aldrin and Dieldrin CSM Summary

Although the production and use of aldrin and dieldrin has been banned since the late 1980s, they have been detected in surface and subsurface sediment and surface water, primarily from RM 6.8W to 7.2W, and dieldrin has also been detected in clams and crayfish samples from this area. This area corresponds to the historical discharge point at RM 6.9 for the former Rhone Poulenc pesticide manufacturing facility. Aldrin and dieldrin have been detected in upland soils and groundwater at this site, and stormwater is a complete historical pathway.

Figures 10.2-19a through 10.2-24 summarize the loading, fate, and transport assessment for aldrin and dieldrin in the Study Area in a typical year. For both chemicals, upriver surface water is the largest estimated external loading term, with the dieldrin surface water loading approximately 50 times higher than that of aldrin. Stormwater is the second highest estimated loading term, followed by atmospheric deposition to the river surface and advection through subsurface sediments.

The BHHRA only identified aldrin and dieldrin as contaminants potentially posing unacceptable risks for fish (carp only for aldrin) and clam consumption based on cancer risk estimates above 1×10^{-6} , but they did not result in cancer risk estimates greater than 1×10^{-4} . Aldrin and dieldrin were identified in the BERA as posing potentially unacceptable risks to the benthic community and sandpiper (based on dietary exposure). The distribution of these contaminants tended to be patchy, and exceedances of sediment thresholds were primarily associated with the west side of the river, below RM 9.1.

10.2.8 Arsenic, Copper, and Zinc

CSMs in a graphical format are presented for arsenic, copper, and zinc on Panels 10.2-9A–C, 10.2-10A–C, and 10.2-11A–C, respectively. All three metals are abundant elements in the earth's crust, and natural releases to environmental media can be significant.

Arsenic is a naturally occurring metal that is found widely in natural minerals, including realgar ($\text{As}_4\text{S}_{4(s)}$), orpiment ($\text{As}_2\text{S}_{3(s)}$), and arsenolite (As_2O_3 ; ATSDR 2005a). It occurs naturally in soil, water, and air as a result of mineral weathering, leaching, volcanic eruptions, and wind-blown dirt (ATSDR 2005). Anthropogenic activities, including smelting, use in pesticides, combustion of wood and coal, waste incineration, and the production and use of treated wood products that utilize soluble chromium copper arsenate (CCA), can also release arsenic into the air, soil, water, and sediments.

Arsenic is a redox-sensitive species, existing at the +3 and +5 oxidation states in aqueous environmental conditions. Under oxidizing conditions the As(V) species (H_3AsO_4 , H_2AsO_4^- , $\text{H}_2\text{AsO}_4^{2-}$, HAsO_4^{2-} , AsO_4^{3-}) predominate, while under reducing conditions the As(III) species (H_3AsO_3 , H_2AsO_3^- , HAsO_3^{2-} , AsO_3^{3-}) predominate (EPRI 1984). Arsenic is generally highly soluble, with few mineral phases exerting controls on aqueous arsenic concentrations under typical environmental conditions. Arsenic sulfide minerals, such as orpiment and realgar, can be formed under reducing and acidic conditions. Although arsenic minerals are generally highly soluble, adsorption reactions to sediment/aquifer mineral grain surfaces frequently limit dissolved arsenic concentrations (Kabata-Pendias and Pendias 1992). Arsenic is particularly strongly adsorbed to iron oxide minerals, with the As(V) species having a greater affinity for the oxide surface than the As(III) species. Because arsenic is frequently present as an anion under typical environmental conditions, its sorption to oxide surfaces is favored at pH <9 (Stumm 1992).

In aquatic environments, bioaccumulation of arsenic occurs primarily in algae and lower invertebrates (ATSDR 2005a). Fish and shellfish can also accumulate arsenic, mainly in the exoskeleton of invertebrates and in the livers of fish. While biomagnification in aquatic food chains is not generally considered significant, predatory fish may biomagnify arsenic through the consumption of prey species (especially bottom dwellers) (ATSDR 2005a).

Copper is an abundant metal element in the earth's crust. Natural releases to environmental media can be significant. Mining operations, agriculture, wastewater sludge, municipal and industrial solid waste, and other industrial processes can also result in environmental releases of copper (ATSDR 2004).

Copper exists in four oxidation states: Cu^0 , Cu^{+1} , Cu^{+2} , and Cu^{+3} (Eisler 1998). Of these oxidation states, the cupric ion (Cu^{+2}) is the most likely to be present in water, though rarely as a free ion. As free ions, cupric ions are the most readily available and toxic inorganic species of copper. However, the cupric ions have a strong tendency to complex or sorb to numerous compounds normally found in natural waters, including suspended solids surfaces and dissolved or particulate organic carbon. Such complexation reduces bioavailability to aquatic organisms (Eisler 1998; EPA 2000a). The amounts of the various copper compounds and complexes present in solution in freshwater depend on water pH, temperature, hardness, and alkalinity; concentrations of bicarbonate, sulfide, and organic ligands; size and density of suspended materials; and rates of coagulation and sedimentation of particulates. Up to 29 different species of copper can be present in aqueous solution in the pH range from 6 to 9. The majority of copper in freshwater from pH 6.0 to 9.3 is in the form of carbonate species (CuHCO_3^+ , CuCO_3 , $\text{Cu}[\text{CO}_3]_2^{2-}$), which have low toxicity (Eisler 1998). Cupric ions account for less than 1 percent of the total dissolved copper in freshwater. Copper carbonate, cupric hydroxide, cupric oxide, and cupric sulfide will precipitate from solution or form colloidal suspensions when excess cupric ions are present (Eisler 1998). The majority of copper released to surface waters settles out or sorbs to sediments (Eisler 1998).

While copper can transform in response to environmental chemistry, it does not degrade.

Copper is taken up by aquatic organisms primarily through dietary exposure and is an essential micronutrient for animals as a component of a number of essential enzymes. Most organisms retain only a small proportion of the copper ingested with their diet. Copper bioconcentrates in aquatic organisms but does not bioaccumulate in mammals or biomagnify in aquatic food chains (EPA 2000a).

Zinc is a common element in the earth's crust and is released to the environment from both natural and anthropogenic sources. Mining and metallurgical processing are the primary anthropogenic sources, along with use of commercial products such as fertilizers and wood preservatives that contain zinc (ATSDR 1997b). Zinc is also used in galvanizing steel and in soldering formulas.

In the environment, zinc occurs as a sulfide, oxide, or carbonate. In freshwater, zinc is most soluble at low pH and low alkalinity: 10 mg Zn/L of solution at pH 6 that declines to 6.5 mg Zn/L at pH 7, 0.65 mg Zn/L at pH 8, and 0.01 mg Zn/L at pH 9 (Eisler 1993). Zinc in the water column¹² can partition to dissolved and particulate organic carbon. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc because the divalent zinc ion is believed to be responsible for observed biological effects (EPA 2000a). Because zinc ligands are soluble in neutral and acidic solutions, zinc is readily transported in most natural waters (Eisler 1993); however, most of the zinc introduced into aquatic environments eventually is partitioned into the sediments (Eisler 1993). Zinc release from sediments is enhanced under conditions of high dissolved oxygen, low salinity, and low pH (Eisler 1993). Zinc may change forms in the environment, but it does not degrade.

Zinc is an essential trace element for all living organisms. As a constituent of more than 200 metalloenzymes and other metabolic compounds, zinc ensures stability of biological molecules such as DNA and of biological structures such as membranes and ribosomes (Eisler 1993). Most studies reviewed contained data that suggest that zinc is not a highly mobile element in aquatic food webs, and there appears to be little evidence to support the general occurrence of biomagnification of zinc within marine or freshwater food webs (EPA 2000a). Bioavailability of zinc in sediments is controlled by the acid-volatile sediment concentration. The arsenic- copper-, zinc-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G).

¹² In water, the free zinc ion is thought to coordinate with six water molecules to form the octahedral aquo ion ($\text{Zn}(\text{H}_2\text{O})_6^{2+}$) in the absence of other complexing or adsorbing agents (Eisler 1993). In typical river waters¹², 90% of the zinc is present as aquo ion, and the remainder consists of ZnHCO_3^+ , ZnCO_3 , and ZnSO_4 (Eisler 1993).

10.2.8.1 Arsenic, Copper, and Zinc Contaminant Distribution

Elevated arsenic concentrations in sediments are defined here as exceeding 20 mg/kg (the orange or red sediment polygons on Panels 10.2-9A–B). Elevated copper concentrations in sediments are defined as exceeding 120 mg/kg (Panels 10.2-10A–B), and elevated zinc concentrations in sediments exceed 400 mg/kg (Panels 10.2-11A–B).

With the exception of a broad area of relatively elevated copper and zinc concentrations in the vicinity of Swan Island Lagoon and zinc at Terminal 4, Slip 3, elevated arsenic, copper, and zinc concentrations in surface and subsurface sediment are generally restricted to small, widely scattered nearshore areas. The similarity of surface and subsurface concentrations in these areas suggests both recent and historical inputs of all three metals.

Total arsenic, copper, and zinc concentrations in surface water were generally consistent across the entire Study Area. Concentrations were generally higher in low-flow sampling events, and there is generally no relationship evident between elevated surface water and elevated surface sediment concentrations. Sediments collected in and upstream of Study Area over the course of a year in sediment traps show little spatial or temporal trends in measured concentrations. The highest reported arsenic concentrations TZW are located at the west side of the channel at RM 6.2–6.6, and the west bank at RM 7.7. However, there are no corresponding high arsenic concentrations in sediment. The highest copper and zinc concentrations in TZW were measured offshore of the Gasco and Siltronic sites in areas where no elevated surface sediment concentrations were reported. Arsenic, copper and zinc were detected in nearly all fish and invertebrate species and tissues analyzed from within the Study Area.

10.2.8.2 Potential Sources of Arsenic, Copper, and Zinc

The areas of elevated sediment concentrations correspond to the locations of some former shipyards, wood treatment facilities, metal recycling operations, pipe manufacturing, metal plating operations, and marine repair facilities. Metals are also associated with some facilities where metal slag and sandblast grit were used as fill. The primary industries in Portland Harbor known to have handled, manufactured, or disposed of arsenic, copper, or zinc include pesticide manufacturing, shipbuilding/demolition and marine repair facilities, metals recycling, battery scrapping, wood treating, and MGP.

A review of the panels reveals that a number of sites with known or likely complete pathways are not associated with proximal elevated surface sediment concentrations. The discussion below focuses on sites with known or likely complete pathways near the larger areas with elevated surface sediment concentrations. As such, sites with known or likely complete pathways not adjacent to these larger areas are not discussed, but are included in the tables and panels.

Known complete or likely complete historical pathways for arsenic, copper, and zinc have been identified at up to 46 sites. A summary of the number of sites with known and likely complete pathways is summarized as follows:

Known or Likely Complete Pathways – Arsenic, Copper, and Zinc

	Groundwater	Stormwater	Overwater	Overland	Riverbank
Arsenic					
Historical	7	35	1	9	13
Current	6	11	0	5	4
Copper					
Historical	6	42	5	9	10
Current	4	13	4	3	3
Zinc					
Historical	4	44	5	9	12
Current	3	12	4	3	3

Although one or more of these metals has been identified as a stormwater COI at sites that drain to a number of municipal and non-municipal shared conveyance systems, associated sediment concentrations are not present near all these outfalls (Tables 10.2-10 through 10.2-12).

Zinc is elevated in sediment in Balch Creek Cove, which is the discharge location for OF-16, OF-17, WR-258, and WR-235. Metals have not been identified as stormwater COIs at WR-258 (drains Fire Station 6) and WR-235 (Port of Portland Terminal 2). GE Decommissioning has a known historical and likely current complete stormwater pathway for zinc. Calbag Nicolai and Galvanizers have likely historical and known current complete stormwater pathways for zinc. All three of these sites are located within the basins of these outfalls.

Arsenic, copper, and zinc concentrations are elevated in sediments offshore of the areas adjacent to Gunderson and the small cove adjacent to the Shaver Transportation and Front Ave LP properties. Gunderson has known complete pathways, both historical and current, for stormwater and overland transport, and Front Avenue LP has a historical known complete stormwater pathway for these metals. Two shared conveyance systems also drain to this area, OF-18 and OF-19. However, none of these metals are elevated in the cove that OF-18 discharges to and only copper is elevated in the vicinity of the OF-19 discharge. Front Avenue LP has a historical likely complete pathway for copper in stormwater and Chevron has a historical known complete pathway for copper in stormwater. Both of these sites drain at least in part to OF-19 (Table 10.2-11).

Elevated sediment concentrations of copper and zinc are present on the nearshore areas on the west side of Swan Island Lagoon and the north end of Swan Island. Cascade General, Swan Island Upland Facility, and Fred Devine are identified as having likely historically complete stormwater pathway for copper and zinc. Single samples of

elevated concentrations of zinc and copper are also present near OFM-1 and of zinc near OFM-2. Freightliner TMP and TMP2 drain to these basins and are identified as a having likely historically complete stormwater pathway for copper and zinc.

Stormwater and groundwater infiltration from sites draining to OF-22B and OF-22C (RM 6.8W) are the likely sources of elevated arsenic concentrations in surface sediment at this location. Stormwater has been identified as a historical and/or current known complete pathway at Rhone Poulenc, Metro Transfer Station, Schnitzer-Doane Lake, Gasco, Siltronic, and Gould, which drained to or have had groundwater infiltration to these outfalls.

TZW sampling results indicate that groundwater is not a significant source to sediments. Low subsurface sediment arsenic concentrations may reflect the dynamic setting—arsenic may be transported downstream before there is an opportunity for long-term burial. Downstream surface sediment concentrations are low, suggesting that if this is the case, that the mass of arsenic being discharged is relatively low and is readily dispersed.

Overland transport has been identified as a likely complete historical pathway for both MarCom parcels, as well as a likely complete current pathway for the North parcel. Elevated surface sediment concentrations are located offshore of the MarCom facilities and surface water concentration immediately upstream appears to support stormwater as a pathway. Calbag Metals, adjacent to the International Slip, has been identified as having known current and historical complete stormwater pathways for zinc and copper; Terminal 4 has a likely historical complete overland pathway for zinc.

Arsenic, copper and zinc are associated with Marine Barge Paint and Blast Area operations at Gunderson and a known historical and current complete pathway for groundwater has been identified for arsenic at Gunderson.

Groundwater is a known or likely current and historically complete pathway for these metals for the Siltronic and Gasco sites. However, none of these metals are present at elevated concentrations in surface or subsurface sediment.

Five sites have been identified as having historical known complete pathways for copper and zinc: Gunderson, Cascade General, Swan Island Upland Facility, McCormick & Baxter, and Schnitzer – Calbag. McCormick & Baxter also has a historical complete pathway for arsenic. All but McCormick & Baxter have known or likely current complete overwater pathways. McCormick & Baxter is the only identified site with a known historical complete pathway for arsenic.

Based on limited riverbank sampling, relatively large areas of elevated metals concentrations in surface sediment are associated with sites with known or likely historical complete riverbank erosion pathways for arsenic, copper, and zinc. These sites include Gunderson, Willamette Cove, MarCom South, and McCormick & Baxter.

A current known or likely complete pathway for riverbank erosion is limited to Gunderson.

There are no known current or historical point sources of atmospheric deposition of these metals in the Study Area. Table 4.5-1 lists upstream ECSI sites (between RM 11.8 and the Tualatin River) and associated COIs.

10.2.8.3 Loading, Fate, and Transport of Arsenic, Copper and Zinc

Loading, fate, and transport of arsenic in the Study Area is summarized on Figures 10.2-25a through 10.2-27. Copper is summarized on Figures 10.2-28a through 10.2-30, and zinc is summarized on Figures 10.2-31a through 10.2-33. Estimated loads from upriver surface water dominate the current loads for all three metals, frequently exceeding the other quantified external loading terms by 2 or more orders of magnitude. Upstream and downstream surface water arsenic loads are comparable. Estimated copper and zinc loads show slightly more entering the Study Area than leaving. The distribution of total arsenic, copper, and zinc (respectively) in surface sediments, sediment trap samples, and the surface water particulate fraction are presented on Figures 10.2-25b, 10.2-28b, and 10.2-31b, summary statistics in each media are presented in Table 10.2-2a and the results of cross-media statistical testing are provided Table 10.2-2b. For each of these metals, all the distributions of each media are significantly different from each other. Concentrations in surface water suspended particles are greater than in sediment trap or surface sediments on a Study Area-wide basis for each metal. Based on median values, sediment trap concentrations are slightly greater overall than surface sediment concentrations for all three metals.

Although the current upstream surface water load estimate greatly exceeds the other current loading terms, there is no indication that this load is responsible for the spatial distribution of the elevated concentrations of these metals observed in Study Area sediments.

10.2.8.4 Relationship of Sources to Distribution of Arsenic, Copper, and Zinc CSM Summary

Elevated arsenic, copper, and zinc concentrations in surface and subsurface sediment are generally restricted to small, widely scattered nearshore areas. The similarity of surface and subsurface concentrations in many of these areas suggests both recent and historical sources. Within the Study Area, numerous historical and current sources have been identified for these metals for all pathways. The areas of elevated sediment concentrations correspond to the locations of some former shipyards, wood treatment facilities, metal recycling operations, pipe manufacturing, metal plating operations, and marine repair facilities. These metals are also associated with some facilities where metal slag and sandblast grit were used as fill.

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Loading, fate, and transport assessments for the Study Area in a typical year indicate that loads from upriver surface water dominate the current loads for all three metals, frequently exceeding the other quantified external loading terms by 2 or more orders of magnitude.

Elevated surface sediment and riparian soil/sediment are present in a small area immediately offshore of OF-48 (RM 7.3, east bank, at the property boundary between McCormick & Baxter and Triangle Park). However, outfall stormwater and solids sampling indicates that the outfall is not a source of arsenic (City of Portland 2010). Although McCormick & Baxter has a historical complete pathway and Triangle Park has a likely historical and known current complete arsenic pathway (Panels 10.2-9A–B), sediment samples collected beneath the McCormick & Baxter cap and upstream offshore of Triangle Park (Panels 10.2-9A–B) have low concentrations, and the single subsurface sediment core does not have elevated concentrations. Along with the apparent depositional setting in this area, the distribution of arsenic indicates relatively recent releases

In the BHHRA, arsenic resulted in cancer risk estimates for fish consumption by an adult tribal fisher that exceeded a cancer risk of 1×10^{-4} . Arsenic was identified as a contaminant potentially posing unacceptable risks for fish and shellfish consumption, direct exposure to in-water sediment, direct exposure to beach sediment, and hypothetical use of untreated surface water as a domestic water source. Copper was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario in the BHHRA. Zinc was identified as a contaminant potentially posing unacceptable risks for fish consumption based on an HQ of 2 for non-tribal child fish consumption, which was based on a single sample of whole-body common carp tissue collected from RM 4 to 8. Zinc was not identified as a contaminant potentially posing unacceptable risks for any other exposure scenario.

Although arsenic, copper, and zinc were identified as posing potentially unacceptable ecological risks, the evidence was inconclusive due to uncertainties and conflicting lines of evidence, and none of these COPCs was recommended to be included as a COC.

10.2.9 Chromium

The Study Area graphical CSM for chromium is presented on Panels 10.2-12A–C. Chromium is a naturally occurring element found in rocks, animals, plants, and soil. It can exist in several different forms in the soil, sediment, water, and air. Chromium(III) occurs naturally in the environment but is also a product of industry. Chromium(0) is used for making steel. Chromium(III) and chromium(VI) forms are produced by the chemical industry and are used for chrome plating, the manufacture of dyes and pigments, leather tanning, and wood preserving. Smaller amounts are used in drilling muds, rust and corrosion inhibitors, textiles, and toner for copying machines (ATSDR 2008).

In the environment, chromium can be found in air, soil, and water. Chromium compounds will usually remain in the air for less than 10 days, depositing to the land and water, especially by wet deposition (ATSDR 2008). Most chromium in water binds to soil and other materials and is subsequently subject to sediment transport processes, though a small amount may dissolve in the water. It can easily change from one form to another in water and soil, depending on the conditions present (ATSDR 2008). The relation between Cr(III) and Cr(VI) in the environment is strongly dependent on pH and oxidative properties of the location, but in most cases the Cr(III) is the dominating species (Kotas and Stasicka 2000).

Although chromium(III) is required in trace amounts for sugar and lipid metabolism in humans and its deficiency may cause a disease called chromium deficiency, chromium(VI) is a toxin and a carcinogen (ATSDR 2008). Fish do not significantly accumulate chromium in their bodies from water (ATSDR 2008).

10.2.9.1 Chromium Contaminant Distribution

For chromium, concentrations greater than 75 mg/kg in sediment, 0.75 µg/L in surface water, 100 µg/L in TZW, and 10 mg/kg in biota samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-12A–C).

Areas of elevated chromium concentrations in surface and subsurface sediments in the Study Area all occur in a few, widely scattered nearshore areas and the head of the International Slip and are limited in spatial extent, and include RM 2E, RM 4E, RM 6E, Swan Island Lagoon, RM 6W, RM 7W, and RM 9W. The distribution of concentrations in surface and subsurface sediments suggest both recent and historical sources.

Sediment trap samples show a uniform distribution of chromium levels seasonally and throughout and upstream of the Study Area (see Figure 5.2-27). Most sediment trap samples fall between 30 and 40 mg/kg; the single highest value (60 mg/kg) was measured in upstream of the Study Area at RM 15.7E during the May to August period.

Chromium was sampled offshore of sites between RM 6.2W and 7.6W. Elevated TZW concentrations (greater than 100 µg/L) were observed between RM 6.2W and 6.5W, offshore of the Gasco and Siltronic properties.

Chromium was detected in all fish and invertebrate species and tissues analyzed within the Study Area.

10.2.9.2 Potential Sources of Chromium and Pathways

In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of chromium include the steel industry (EOSM), ship building/demolition and marine repair facilities (Gunderson, Cascade General, MarCom), metal recycling

(Schnitzer-Calbag, Calbag-Nicolai, Calbag Metals-Front Ave., and former operations at Schnitzer-Doane Lake), wood treating (McCormack & Baxter), and MGP sites (Gasco), and heavy oil facilities (bulk fuel and asphalt storage). Sodium bichromate was used in the sodium chlorate manufacturing process as a corrosion inhibitor at Arkema, however it is highly soluble and unlikely to be found in sediments.

Historical known complete or likely complete pathways for chromium have been identified at 43 sites (Table 10.2-13 and Panels 10.2-12A–C), and include stormwater (38 sites), groundwater (seven sites), overwater releases (five sites), overland transport (nine sites), and riverbank erosion (10 sites). Current known complete or likely complete pathways for chromium have been identified at 20 sites and include stormwater (13 sites), groundwater (five sites), overwater releases (four sites), overland transport (three sites), and riverbank erosion (three sites).

Groundwater is a historical known or likely complete pathway for chromium at seven sites, including two which are not current sources (McCormick & Baxter and Schnitzer-Calbag). Elevated sediment concentrations near OF-22B and OF-22C may be related to sites with known or likely complete groundwater / stormwater infiltration draining to these outfalls. Bulk fuel and MGP facilities with complete stormwater pathways (Table 10.2-13) are not in areas of elevated sediment concentrations.

Based on limited riverbank sampling, riverbank erosion is a historical known or likely complete pathway for chromium at 10 sites, including six which are not current sources. Of the 10 sites, elevated sediment concentrations are near steel manufacturing operations (EOSM), metals recycling (Schnitzer-Calbag), and two shipyards (Portland Shipyards and Gunderson).

Chromium was identified as a COI for 17 upstream sites based on their hazardous substances/waste types. Of these 17 sites, chromium was detected in sampled media at the following five sites:

- Willamette Falls Locks – Chromium detected in upland soil, and direct releases/spills represent potential pathways to the river
- Zidell Marine Corporation – Chromium detected in upland soil, and direct releases/spills, groundwater, and stormwater represent potential pathways to the river
- OHSU - Moody Ave. Units A, B, C – Chromium detected in upland soils, and stormwater and groundwater represent potential pathways to the river
- Clackamette Cove Area – Chromium detected in upland soils, and groundwater represents a potential pathway to the river
- South Waterfront Redevelopment Area 3 – Chromium detected in groundwater, which represents a potential pathway to the river.

10.2.9.3 Loading, Fate, and Transport of Chromium

Chromium loading, fate, and transport assessment for the Study Area is summarized on Figures 10.2-34a through 10.2-36. Estimated loads from upriver surface water dominate the current loads, exceeding the next highest external loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. The chromium loads in upstream surface water and the downstream boundary of the Study Area are generally comparable, reflecting the absence of significant loads within the Study Area. The nearshore and off-channel areas of elevated chromium concentrations generally appear to be associated with localized upland sources and pathways.

The cross-media comparison and statistical assessment of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-34b) show that the Study Area-wide concentrations of all media are statistically different, with surface sediment having the highest concentrations, followed by sediment traps and then by suspended solids in surface water.

10.2.9.4 Relationship of Sources to Distribution of Chromium CSM Summary

Within the Study Area, numerous historical and current sources of chromium have been identified for all pathways, but primarily through stormwater discharge. The areas of elevated sediment concentrations generally correspond to the locations of current or former shipyards, wood treatment facilities, pesticide manufacturing, metal recycling operations, steel manufacturing, metal plating operations, and marine repair facilities. Chromium is also identified at some facilities where metal slag and sandblast grit were used as fill. Known and likely current and historical sources of chromium to the Study Area are summarized in Table 10.2-13 and Panels 10.2-12A–C.

The chromium loading estimates indicate that loads from upriver surface water dominate the current loads, exceeding the next highest external loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. However, the estimated loads entering and leaving the Study Area are comparable, suggesting the absence of significant loads within the Study Area.

Total chromium was not identified as a contaminant potentially posting unacceptable risks for any exposure scenario in the BHHRA. Hexavalent chromium was identified as a contaminant potentially posing unacceptable risks for the hypothetical use of untreated surface water as a drinking water source based on a single location (RM 7.0). The BERA identified chromium as posing potentially unacceptable risk to benthic invertebrates, but exceedances of SQVs were of low magnitude and limited areal extent.

~~40.2.44~~10.2.10 Tributyltin Ion

The CSM for TBT is presented on Panels 10.2-13A–C. TBT is an organotin compound, and since the mid-1970s has been and is still used as an antifouling agent in paints on the immersed portions of boats and floating structures (Batt 2004). Antifouling paints represent the largest source of TBT in coastal environments, and nationally. Many countries restricted the use of antifouling paints based on the risks to shellfish. The U.S. partially banned the use of TBT-based antifouling paints in 1988 (Showalter and Savarese 2005). Use of TBT compounds as slimicides on masonry, disinfectants, and biocides for various industrial processes also may result in their release to the environment.

TBT is an ionic organic compound, and its partitioning behavior is affected by pH and the identity of anions in solution that pair with the TBT ion (Arnold et al. 1997). Specifically, for pH 10 to 7 the measured log K_{oc} values are on the order of 4; from pH 7 to pH 3 they drop to roughly 2. The mean surface water pH in Portland Harbor is 7.4 (10th percentile is 7.0 and 90th percentile is 7.8). Observed pore water pH values ranged from 5.6 to 8.1. TBT in its nonionic form (not likely observed at the site) is highly hydrophobic, with K_{ow} values on the order of 7 (K_{ow} WIN¹³).

Degradation of organotin compounds involves the breaking of the tin-carbon bond. TBT is subject to different degradation mechanisms depending on its location in the environment. In surface water, it is subject to fairly rapid¹⁴ photodegradation and biodegradation (TBT can be degraded by microbial, microalgal, and fungal populations, as well as by some higher organisms, such as fish; Anderson et al. 2002). Degradation of organotin compounds in sediments is much slower than in water, and half-lives have been estimated to be several years (Alzieu 1998). Abiotic cleavage of the tin-carbon bond by hydrolysis is not an important fate process under environmental conditions (WHO 1990).

While accumulation of TBT from water into organisms may be a significant process that can result in elevated tissue concentrations, subsequent biomagnification through the food web is reportedly minor (ATSDR 2005b).

¹³ K_{ow} WIN software available online as part of EPA Estimation Program Interface (EPA) suite of programs: <http://www.epa.gov/oppt/exposure/pubs/episuite.htm>

¹⁴ Half-life information for TBT in freshwater could not be found; however, the following information was found for seawater: the half-life of tributyltin in seawater varies, depending on pH, temperature, turbidity, and light; it is generally estimated to be in the range of 1 day to a few weeks (Alzieu 1998). Biodegradation is the major process in seawaters rich in suspended solids, but photolysis, in surface waters, exceeds biodegradation in clean seawater. Calculated half-lives range from 6 days in summertime waters rich in suspended particles to 127 days in clean winter waters (Watanabe et al. 1992).

10.2.11.10.2.10.1 TBT Contaminant Distribution

For TBT, concentrations greater than 1,000 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 0.01 µg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-13A–C).

TBT contamination in sediment is primarily located in the vicinity of the Cascade General Shipyard and adjacent to Swan Island Lagoon. Concentrations in subsurface sediments exhibit slightly higher concentrations than surface sediments, suggesting that contributions from historical inputs were greater relative to current inputs. TBT was not detected in sediment trap samples upstream of RM 7.5 (with the exception of Swan Island Lagoon).

10.2.11.210.2.10.2 Potential Sources and Pathways of TBT

Within the Study Area, historical and current sources contributed TBT to the river primarily through the overwater and stormwater pathways. Areas of elevated TBT concentrations correspond with current and former shipyards, where ship hull washing, abrasive blasting, and painting occurred in dry docks and berths (see Map 3.2-3).

Identified known and likely current and historical sources of TBT to the Study Area are summarized in Table 10.2-14 and Panels 10.2-13A–C. Historical pathways for TBT migration are found at four current and historical shipyard facilities (Cascade General, Gunderson, Mar Com South, and Marine Finance) and include stormwater (four sites) and overwater, overland, and riverbank erosion (two sites). Current known complete or likely complete pathways for TBT have been identified at two sites (Cascade General and Gunderson).

A current and historical known complete stormwater pathway to the river exists at Gunderson, an active manufacturer and refurbisher of railroad cars and marine barges since 1913 (Integral 2007a). Historical likely complete stormwater pathways also exist at Cascade General, MarCom South, and Marine Finance. Historical overland transport pathways are likely complete at MarCom South and Marine Finance. The overwater discharge pathway is considered to be the greatest contributor of TBT to the river, current and hHistorical complete pathways for overwater discharge of TBT are found at Cascade General and a complete historical overwater pathway existed at MarCom South. There are no current or historical groundwater pathways for TBT in the Study Area. Likely complete historical pathways for riverbank erosion have been identified at MarCom South and Marine Finance. Information on atmospheric deposition of TBT is very limited, but long-range atmospheric transport of butyltins does occur.

TBT was identified as a COI at two upstream sites, Zidell Marine Corporation site (RM 14W) and Ross Island Sand & Gravel (RM 14.7E), based on detections in sampled media.

~~10.2.11.3~~ 10.2.10.3 Loading, Fate and Transport of TBT

The loading, fate, and transport assessment for TBT in the Study Area is summarized on Figures 10.2-37a through 10.2-39. Estimated TBT inputs from upriver surface water (11 kg/yr) are lower than those from advection through subsurface sediments (36 kg/yr). The relatively high overall subsurface advective load estimate is driven by localized TBT concentrations at RM 8 to 8.9. Quantitative estimates of internal fate and transport processes were developed only for advection through surface sediments (9.8 kg/yr), which comparable in magnitude to upstream surface water loading. Advective loading for both pathways is greatest between RM 8 and 8.9 (Figure 10.2-39).

Cross-media comparison and statistical assessment of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-37b) show that the concentrations in all media are statistically different.

~~10.2.11.4~~ 10.2.10.4 Relationship of Sources to Distribution of TBT CSM Summary

Since 1988, TBT has been partially banned in the U.S. for use as an ingredient in anti-fouling paint. Active shipyards in the Study Area have since adopted alternative paints and abrasive blasting methods as well as BMPs that will lessen the discharge of TBT to in-water media. TBT has been detected in surface and subsurface sediment, and fish and invertebrate samples in the vicinity of the Cascade General Shipyard and adjacent to Swan Island Lagoon. At the Cascade General Shipyard, concentrations in subsurface sediments are greater than in surface sediment. Surface sediment and riparian soil samples are elevated where OF-18 empties into the river on the Gunderson shoreline. OF-18 drains stormwater from Gunderson as well as several other ECSI sites within its basin. However, TBT is a COI only at Gunderson.

As shown in Figures 10.2-37a through 10.2-39, advection through subsurface sediments appears to be the largest loading term, followed by upriver surface water. The relatively high overall subsurface advective load estimate is driven by localized elevated TBT concentrations at RM 8 to 8.9.

In the BHHRA, TBT was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario. In the BERA, TBT was identified as posing potentially unacceptable risk to fish and benthic invertebrates based on surface water, sediment, and empirical and predicted tissue residue lines of evidence. Potential risks to fish and invertebrates are highly localized in historical shipyard areas.

~~10.3~~ CONCLUSIONS

~~The key findings of the RI include the following:~~

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Sources of Contamination

Most of the sediment contamination at the Site is associated with known or suspected historical sources and practices. ongoing contaminated groundwater plumes, river bank soils, and upstream surface water. The distribution of contaminants in sediments in several nearshore areas appears to reflect more significant historical lateral inputs. The spatial correlation between PCBs in aquatic organisms and sediments indicate that contamination in bottom sediments are an ongoing source of persistent bioaccumulative contaminants such as PCBs, DDT, and dioxin/furans contamination to biota.

The major internal fate and transport processes are:

- Erosion from the sediment bed
- Deposition to the sediment bed
- Dissolved flux from the sediment bed (porewater exchange)
- Groundwater advection
- Degradation (for some contaminants)
- Volatilization
- Downstream transport of either particulate-bound or dissolved-phase contaminants

These processes interact to create complex patterns of contaminant redistribution that vary over space, time, and by contaminant. Patterns of contamination in bedded surface sediment suggest some redistribution of contaminants over time from past source areas, but this is limited by re-burial of much of the source area contamination (as indicated by higher subsurface sediment concentrations in these areas). Periodic erosion may temporarily expose buried contamination.

Groundwater plume advection and release has been observed in several areas along with dissolved phase flux from surface sediments to the water column.

Based on results of surface water data collected during the RI, resuspension and/or dissolved phase flux from the sediment bed are contributing to contaminant concentrations in surface water, particularly in quiescent areas where surface water mixing and dilution is minimal. Loading estimates are consistent with this concept, indicating the mass flux of contaminants exiting the downstream end of the Study Area in surface water (either directly to the Columbia River or via Multnomah Channel) is greater than the flux entering the Study Area. Contaminant concentrations in loads entering the Study Area from adjacent upland sources and pathways (such as stormwater) are generally greater than concentrations in the upstream loads (upriver surface water and sediments). Stormwater input is the most important current source pathway within the Study Area for many contaminants, including PCBs and DDT.

Finally, empirical tissue contaminant data and food web modeling indicate that persistent contaminants (particularly PCBs and dioxin/furans) in sediments and surface water bioaccumulate in aquatic species tissue. There is little evidence from

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concentration gradients in bedded surface sediment data of significant migration of contaminants immediately downstream of the Study Area in either the Willamette River main stem or Multnomah Channel.

Nature and Extent of Contamination

- Sediments in Portland Harbor reflect the industrial, marine, commercial, and municipal practices for over 100 years in this active industrial, urban, and trade corridor, as well as agricultural activities in the Willamette Basin.
- Higher concentrations of contaminants in sediments occur in nearshore and off-channel areas that are generally associated with known or likely historical or current sources.
- Contaminant concentrations in sediment are generally higher at depth than in the surface layer, indicating that past contaminant inputs were greater than current inputs, and that surface sediment quality has improved over time. The few exceptions include areas where higher surface sediment concentrations appear to be associated with ongoing local sources, low rates of sediment deposition, and physical sediment disturbance (e.g., from boat scour).

Estimates of Risk

The major findings of the BHHRA are:

- Estimated cancer risks resulting from the consumption of fish or shellfish are generally orders of magnitude higher than risk resulting from direct contact with sediment and surface water. Consumption of resident fish species consistently results in the greatest risk estimates. Evaluated harbor wide, the estimated RME cancer risks are 4×10^{-3} and 1×10^{-2} for recreational and subsistence fishers, respectively. Noncancer hazard estimates for consumption of resident fish species are greater than 1 at all river miles. The highest noncancer hazards are associated with nursing infants of mothers, who consume resident fish from Portland Harbor. When fish consumption is evaluated on a harbor wide basis, the estimated RME HI is 4,000 and 10,000 for infants of recreational and subsistence fishers, respectively. Evaluated on a harbor wide scale, the estimated RME HI for tribal consumers of migratory and resident fish is 600 assuming fillet-only consumption, and 800 assuming whole body consumption. The corresponding HI estimates for nursing infants of mothers, who consume fish, are 8,000 and 9,000 respectively, assuming maternal consumption of fillet or whole body fish.
- PCBs are the primary contributor to risk from fish consumption harbor wide. When evaluated on a river mile scale, dioxins/furans are a secondary contributor to the overall risk and hazard estimates, particularly at RM 6 and 7. PCBs are the primary contributors to the noncancer hazard to nursing infants, primarily

because of the bioaccumulative properties of PCBs and the susceptibility of infants to the developmental effects associated with exposure to PCBs.

The following presents the primary conclusions of the BERA.

- In total, 93 contaminants (as individual contaminants, sums, or totals) pose potentially unacceptable ecological risk. The list of contaminants posing potentially unacceptable risks can be condensed if individual PCB, DDx and PAH compounds or groups are condensed into three comprehensive groups: total PCBs, total DDx, and total PAHs. Doing so reduces the number of contaminants with $HQ \geq 1.0$ posing potentially unacceptable risks to 66.
- Risks to benthic invertebrates are clustered in 17 benthic AOCs. Sediment and TZW samples with the highest HQs for many contaminants also tend to be clustered in areas with the greatest benthic invertebrate toxicity. The COPCs in sediment that are most commonly spatially associated with locations of potentially unacceptable risk to the benthic community or populations are PAHs and DDx compounds.
- Not all COPCs posing potentially unacceptable risk have equal ecological significance. The most ecologically significant COPCs are PCBs, PAHs, dioxins and furans, and DDT and its metabolites. The contaminants identified as posing potentially unacceptable risk in the largest numbers of LOEs are (in decreasing frequency of occurrence) total PCBs, copper, total DDx, lead, tributyltin (TBT), zinc, total toxic equivalent (TEQ), PCB TEQ, benzo(a)pyrene, cadmium, 4,4'-DDT, dioxin/furan TEQ, bis(2-ethylhexyl) phthalate, naphthalene, and benzo(a)anthracene. The remaining 78 contaminants posing potentially unacceptable risk were identified as posing potentially unacceptable risk by three or fewer LOEs.
- Of the three groups of contaminants (i.e., total PAHs, total PCBs, total DDx) with the greatest areal extent of $HQs \geq 1.0$ in the Study Area, PAH and DDx risks are largely limited to benthic invertebrates and other sediment-associated receptors. PCBs tend to pose their largest ecological risks to mammals and birds.

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Commented [A18]: For Eco Risk, PAH risk should be described in terms of Total PAH, Total HPAH or Total LPAH instead of benzo(a)pyrene.